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THE BALANCES OF ^{137}Cs, STABLE CESIUM, AND THE FEEDING RATES OF BLUEGILL (LEPOMIS MACROCHIRUS RAF.) IN WHITE OAK LAKE

(Thesis)

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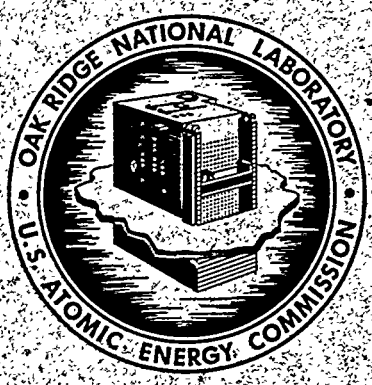

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THE BALANCES OF ^{137}Cs, STABLE CESIUM, AND THE FEEDING RATES OF BLUEGILL (LEPOMIS MACROCHIRUS RAF.) IN WHITE OAK LAKE

S. E. Kolehmainen and D. J. Nelson

Submitted as a dissertation by S. E. Kolehmainen to the Graduate Council of the University of Tennessee in partial fulfillment of the requirements for the degree of Doctor of Philosophy

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

FIGURE PAGE

PREFACE

Since the beginning of the nuclear weapons testing in 1945, fallout radionuclides have been distributed all over the world from the highest mountain top to the depths of sea. In addition to this worldwide fallout, nuclear power plants, nuclear vessels, and isotope laboratories release more localized radioactive pollution chiefly into aquatic environments.

Cesium-137 is one of the most dangerous radionuclides in fallout and also in the effluents of nuclear power plants because of its long physical half-life and its tendency to be concentrated in organisms including man. The intensive nuclear bomb testing in the early sixties produced fallout ^{137}Cs in great quantities. Even though the major tests ended in 1962, the highest concentrations in organisms occurred one to three years later because accumulation in food chains is delayed. Fallout ^{137}Cs has been detected in all the foodstuffs and in all people, but because of certain ecological factors the concentration of ^{137}Cs was especially high among nomadic populations in arctic and subarctic regions, viz, among Lapps and Alaskan Eskimos (Liden and Gustafsson, 1967; Miettinen et al. 1963, 1967; Hanson et al. 1964). The high body burden of these people was a result of the food chain: lichen - reindeer (or caribou) - man. Lichens have a long lifespan and ^{137}Cs is concentrated efficiently on their large surface area. Reindeer eat lichen as their only food item in winter, and reindeer meat, which is the main food item of Lapps, reached high concentrations of ^{137}Cs during 1963-66 (Miettinen, 1967).

Eskimos and Lapps also eat considerable quantities of freshwater fish, especially in summer, when fish are the main source of ^{137}Cs. In arctic and subarctic regions lakes are of the oligotrophic type with a low concentration of potassium. In this type of lake fish accumulate ^{137}Cs efficiently through their food chains (Hä̈sä̈nen and Miettinen, 1963; Kolehmainen et al. 1964, 1966, 1967, 1968b,c, Liden, 1961, 1964). Hence, people eating freshwater fish as well as reindeer are potential accumulators of large quantities of ^{137}Cs fallout in subarctic regions. Because of health physics and ecological viewpoints, the factors affecting the accumulation of ^{137}Cs in fish have been studied widely (Ichikawa, 1960; Williams and Pickering, 1961; Baptist and Price, 1962; Feldt and Lange, 1962; Feldt, 1963, 1966; King, 1964; Kolehmainen et al. 1964, 1967, 1968a, b, c; Kevern et al. 1964; Lebedeva, 1966a, b; Bortoli et al. 1967b; Nelson et al. 1967; Preston et al. 1967; Nelson, 1969).

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ABSTRACT

The concentration of ^{137}Cs, stable cesium and potassium and the specific activities of ^{137}Cs were determined in seven species of fish in White Oak Lake, a small radioactively contaminated lake in Oak Ridge, Tennessee, during a period from June of 1967 to January of 1969. The intake of ^{137}Cs was used to calculate the feeding rates of bluegill at different times of the year.

The concentration of ^{137}Cs in bluegill and other species of fish as well as in the food of bluegill was analyzed γ-spectrometrically. Concentrations of stable cesium and potassium were analyzed by flame spectrophotometry after a chemical separation. The concentration of ^{137}Cs in bluegill increased by a factor of four in fish from 1 to 70 g, but the concentration of stable cesium decreased somewhat with increasing fish size. In bluegill above 70 g the concentration of ^{137}Cs was in a steady state with a considerable seasonal cycling, and the concentration of stable cesium in large bluegill had a seasonal cycling similar to that of ^{137}Cs. Concentrations of potassium also decreased slightly with increasing fish size, but the concentration of potassium did not fluctuate seasonally.

The excretion rates of cesium and potassium were determined by biological half-life studies using ^{137}Cs and ^{42}K. The biological half-life of ^{137}Cs increased with the increasing size of bluegill. There were no significant differences in the long component of the biological half-life of ^{137}Cs among fish that were close to equilibrium with ^{137}Cs and fish that received a single feeding of ^{137}Cs. The short component of the biological half-life of ^{137}Cs was 8 days and the long

component 190 days at 15 C. The biological half-life of potassium consisted of only one component which was 40 days at 15 C. The assimilation of ^{137}Cs increased with fish size and it varied for different types of food from 3% to 70%. The assimilation of potassium was almost complete for all types of food.

The intakes of ^{137}Cs, stable cesium, and potassium in bluegill were calculated for a non-equilibrium state considering the growth of the fish, the seasonal cycling of the concentration and the excretion rates. Feeding rates were calculated on the basis of the daily ^{137}Cs intake, the concentration of ^{137}Cs in bluegill's food, and the assimilation percentage of ^{137}Cs for different food items. The feeding rate of bluegill was in a positive relationship with the temperature of water, but the correlation was not complete. The daily meal was at a minimum, 0.8% of body weight, in February increasing rapidly during May to a maximum, 3.2% in June whereafter it decreased gradually towards the winter. The annual mean of daily meal was 1.75% of body weight. The annual mean of the feeding rates of bluegill calculated on the basis of stable cesium and potassium balances agreed well with the mean of feeding rates calculated by the ^{137}Cs balance.

The concentration of ^{137}Cs was highest in golden shiner and lowest in redear sunfish while the concentration of stable cesium was highest in goldfish and lowest in bluegill. The concentration of potassium varied only slightly among different species of fish. There were no consistent relationships between ^{137}Cs, stable cesium, and potassium concentrations in different species of fish in White Oak Lake. The specific activity of ^{137}Cs in different fish species varied from one to two and a half times that of water.

CHAPTER I

INTRODUCTION

Physiological studies have shown that cesium, rubidium, and potassium have many biological properties in common (Relman, 1956). All these elements belong to Group I alkali metal series in the Periodic Table of Elements. This group has the order: lithium, sodium, potassium, rubidium and cesium. Potassium, rubidium and cesium have very similar electrical, chemical and physical properties while lithium and sodium have more in common with each other. The proportions of cesium, rubidium and potassium in igneous rocks are 0.0007%, 0.031%, 2.59%, respectively (Rankama and Sahama, 1950). The interactions among radiocesium, stable cesium, and potassium are poorly known in organisms. It has been shown, however, that because of the relatively greater quantity of potassium compared to stable cesium, potassium in water plays an important role as a nonisotopic carrier for radiocesium in aquatic organisms (Nelson, 1960; Bryan, 1963a,b; Bryan et al. 1966; Friend et al. 1965; Beninson et al. 1966; Kolehmainen et al. 1966, 1967, 1968a,b, c).

White Oak Lake is a small reservoir that receives a chronic input of radioactive wastes from Oak Ridge National Laboratory. The concentration of radiocesium in water does not fluctuate greatly and the biological components of the ecosystem should be able to reach an equilibrium with the ^{137}Cs in water. Under these circumstances White Oak Lake gives an ideal opportunity to study the balance of ^{137}Cs in fish.

Organisms are not able to distinguish between radiocesium and stable cesium and therefore the ratio between the concentrations of radiocesium and stable cesium, so called specific activity, would be the same in water and in fish in an equilibrium state if radiocesium and stable cesium are in the same physico-chemical state. Cesium and potassium both are taken via the gastrointestinal system in animals, and both are concentrated by muscle tissues. In spite of these similarities the radiocesium to potassium ratio is not as useful for predicting the concentrations of radiocesium in organisms as the radiostrontium to calcium ratio for predicting the radiostrontium concentrations (Comar and Wasserman, 1960; Kornberg, 1960; Nelson, 1969). Knowledge of the relationship between cesium and potassium in fish is limited, and one of the main reasons is the lack of data on the rates of intake and excretion of potassium in fish.

The objectives of this study were to analyze the factors that determine the balances of cesium and potassium in bluegill. These factors are: (1) intake, (2) body burden, and (3) excretion. If two of the factors are known, the third can be calculated. Intake is the most difficult to determine under field conditions, and so in this study the intake was calculated on the basis of the body burdens and excretion rates. Since fish assimilate cesium and potassium from food the daily intake of these elements could be utilized to calculate the daily food consumption at different times of the year.

The validity of the specific activity concept for ^{137}Cs in White Oak Lake was tested by analyzing the concentrations of ^{137}Cs and stable cesium in different species of fish and in water.

CHAPTER II

LITERATURE REVIEW ON POTASSIUM, STABLE CESIUM AND ^{137}CS IN AQUATIC ECOSYSTEMS

Information on the behavior of stable cesium in aquatic organisms is very limited. This is expected since cesium is a rare element without any known physiological response in organisms (Davis, 1963). On the other hand the limited knowledge of potassium in aquatic ecosystems is surprising because potassium is one of the major nutrients in organisms.

Potassium and Stable Cesium in Aquatic Environments

The concentration of potassium in sea water is 380 mg/ℓ and that of stable cesium ranges from 0.2 to 1.3 µg/ℓ (Yamagata, 1962; Cutshall and Osterberg, 1964; Folsom et al. 1964; Michon, 1964; Polikarpov, 1966; Robertson et al. 1968). In fresh water both potassium and cesium concentrations vary greatly. The concentrations of potassium range from 0.2 to 1.0 mg/ℓ in oligotrophic lakes and in eutrophic lakes the concentrations range from 1 to 10 mg/ℓ (Järnefelt, 1958). Only a few values are reported for stable cesium in fresh waters. According to Yamagata (1951, cited by Livingston, 1963) cesium concentrations in seven rivers in Japan varied from 0.05 to 0.2 ug/ℓ. In a preliminary analysis Bortoli et al. (1969) found stable cesium values from 0.043 to 0.087 µg/ℓ in four northern Italian lakes. The Clinch River near the confluence of White Oak Creek (Clinch River Mile 20.8) contained 0.025 µg/ℓ cesium (Nelson, 1969). In 14 Finnish lakes the concentration of stable cesium varies from 0.1 to 1.2 µg/ℓ (unpublished data) and the concentrations seemed to be lowest in turbid eutrophic lakes and

in oligotrophic lakes while the lakes in the middle of the limnological trophic scale had the highest concentrations. Low concentrations of stable cesium in lakes turbid with clay particles are probably caused by the removal of cesium through sorption by suspended clay particles and clay-containing bottom sediments. With ^{137}Cs it has been shown that the removal rate of cesium is much faster in a eutrophic, turbid lake than in an oligotrophic lake (Kolehmainen et al. 1968a). The great ability of clay to sorb ^{137}Cs has been shown by many studies (Tamura and Jacobs, 1960; Garder and Skulberg, 1964; Shih and Gloyna, 1968).

Potassium is one of the chief intracellular cations, and its abundance in the cell is related to its abundance in the earth's crust (Relman, 1956). Potassium concentrations vary somewhat among different aquatic organisms when calculated on fresh weight basis, but they are fairly uniform if calculated on dry weight basis. Dry weight more nearly reflects the weight of protoplasm in aquatic organisms than fresh weight because these organisms contain variable quantities of water. Potassium concentrations in some marine and fresh water organisms are listed in Tables 1 and 2. The differences in potassium contents between species from sea and fresh water are not great even though the potassium concentration is several orders of magnitude higher in sea water than in fresh water. In fact, the concentration factor (C.F.) of potassium in fish

$$C.F. = \frac{\text{conc. of element in the organism per unit weight}}{\text{conc. of element in water per unit weight}}$$

is inversely correlated with the potassium concentration in water (Fig. 1). (See also Preston et al. 1967.)

TABLE I

POTASSIUM AND STABLE CESIUM CONCENTRATIONS IN SOME MARINE ORGANISMS

	K g/kg	Cs µg/kg	Reference
Marine Algae (non-calcareous)	9.5	0.5	Krumholz et al. 1957
Marine Algae (non-calcareous)	8	5	Michon, 1964
Seaweed	1.7 - 8.6	9 - 120	Bryan et al. 1966
Zooplankton	3	20	Krumholz et al. 1957
Zooplankton		9	Michon, 1964
Molluscs (no shell)	3.1 - 3.9	11 - 14	Bryan et al. 1966
Amphipods		3.6	Robertson et al. 1968
Decapod Muscles	3.3 - 3.7	21.1 - 22.9	Polikarpov, 1966
Decapod Muscles	3.9 - 7.4	11.0 - 13.0	Bryan et al. 1966
Fish Muscles	3.3 - 6.4	10.7 - 17.2	Bryan et al. 1966
Fish Muscles	2.6 -13.3	13 - 122	Bryan et al. 1966
Fish Whole		19 - 25	Fukai and Yamagata, 1962
Fish Whole	3	20	Michon, 1964
Water	380	0.2 - 1.3	Yamagata 1962; Polikarpov, 1966

All values are based on fresh weight.

TABLE 2

POTASSIUM AND STABLE CESIUM CONCENTRATIONS IN SOME FRESH WATER ORGANISMS

	K g/kg	Cs µg/kg	Reference
Higher Plants	1.5 - 3.0		Kolehmainen et al. 1968c
Zooplankton	0.17 - 1.26		Kolehmainen et al. 1968c
Bottom Animals	1.0 - 2.7		Kolehmainen et al. 1968c
Fish	2.0 - 4.0		Kolehmainen et al. 1966
Fish	3.0 - 3.43		Bortoli et al. 1969
Fish Flesh	3.26 - 3.52	3.4 - 16.0	Nelson, 1969
Water, Lakes in Finland	$0.2 - 3.5 \cdot 10^{-3}$		Kolehmainen et al. 1966
Water, Lakes in Northern Italy	$1.0 - 2.0 \cdot 10^{-3}$	0.043 - 0.087	Bortoli et al. 1969
Water, Clinch River	$1.3 \cdot 10^{-3}$	0.028	Nelson, 1969

All values are based on fresh weight.

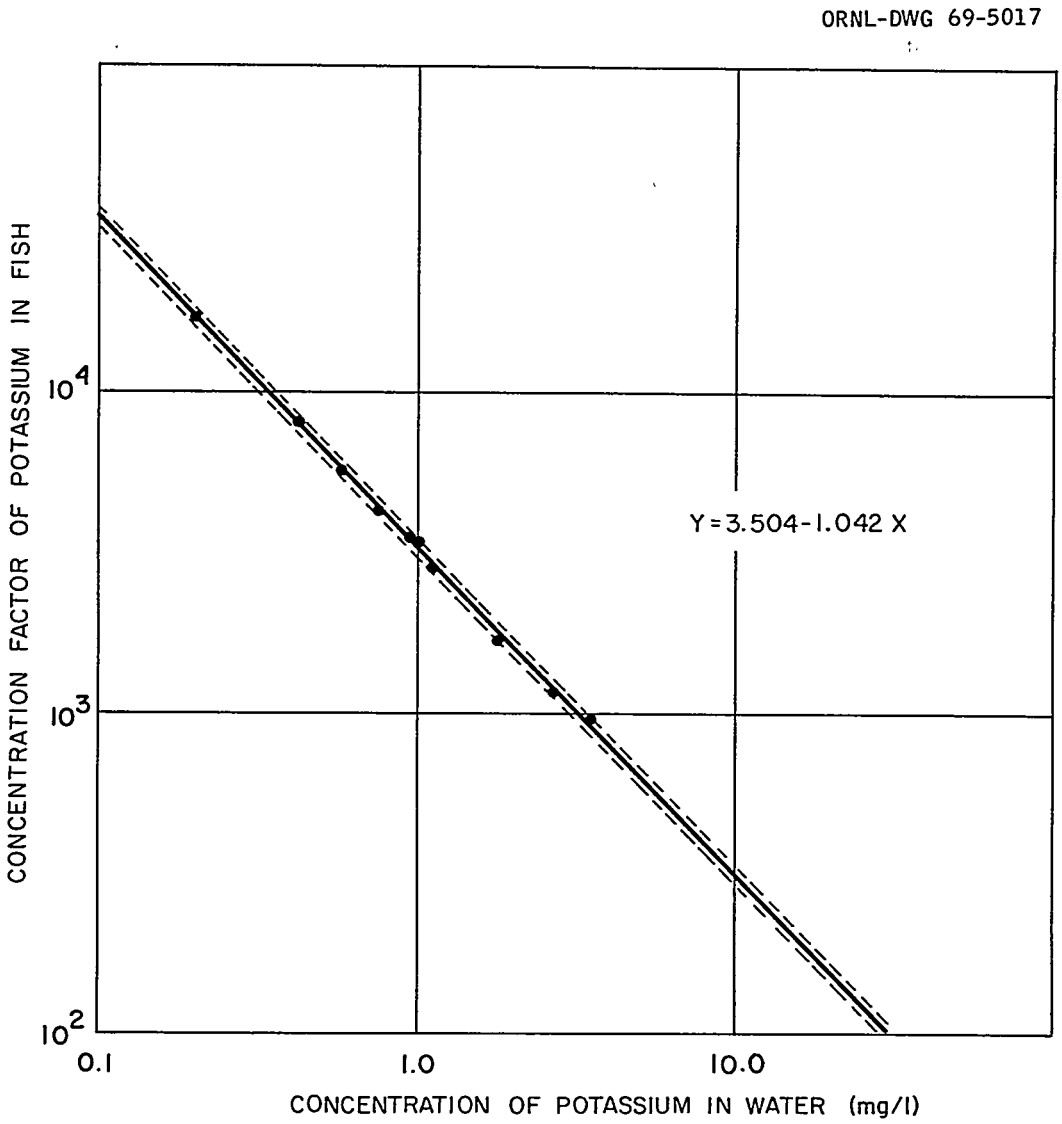

Figure 1. Concentration factors of potassium in northern pike (Esox lucius) as a function of the concentration of potassium in water.

Data taken from Kolehmainen et al. 1968b.

The cesium content of marine species is known somewhat better than in fresh water species (Tables 1 and 2). Among fresh water organisms fish are the only group that have been studied for stable cesium. Nelson (1969) reported cesium values from 3.4 to 16.0 µg/kg fresh weight in the flesh of five species in Clinch River and from 7.6 to 14.4 µg/kg fresh weight in the whole fish of four species in White Oak Lake. Surprisingly both potassium and cesium concentrations in marine fish and Clinch River fish are of the same level. However, the fish in northern Italian lakes seem to have somewhat higher concentrations of stable cesium (9.8 - 57 µg/kg) (Bortoli et al. 1969) and it appears from their data that the concentration of stable cesium in fish might be directly related to the concentration of stable cesium in water and inversely related to the concentration of potassium in water. Concentration factors of stable cesium in three species of fish in four Italian lakes ranged from 110 - 1100. For some reason the C.F.'s of ^{137}Cs in fish were four times higher than those of stable cesium (Bortoli et al. 1967b).

Fate of ^{137}Cs in Aquatic Environments

Sources of ^{137}Cs in Aquatic Environment

The fissioning of uranium and plutonium produces about 6% of fission product ^{137}Cs. This radioisotope is one of the most common fission products, and because of its long half-life, 30 years, it contaminates the environment for hundreds of years. In an aquatic environment the main sources of ^{137}Cs are

weapon testing fallout, nuclear power plants, nuclear vessels and radioisotope laboratories.

The quantity of ^{137}Cs in the aquatic environment has fluctuated considerably depending on the extent of the atmospheric weapons tests (Bortoli et al. 1964, 1965a,b, 1966, 1967a; Salo, 1966, 1967; Gustafson, 1969). After the nuclear test ban treaty became effective (1962), the global fallout produced has been negligible. The half-time of fallout in stratosphere is slightly more than two years, and consequently the corresponding concentrations of fallout ^{137}Cs organisms lag from 1 to 3 years behind nuclear explosions (Kolehmainen et al. 1967, 1968b, c; Preston et al. 1967; Gustafson, 1969). Seasonal maximum of fallout deposition is in spring in the northern hemisphere (Kuroda et al. 1959; Comar, 1963; Lavrenchik, 1965; Bortoli et al. 1964, 1965a, b; 1966, 1967a). If ^{137}Cs fallout comes into contact with soil, most of the radionuclide is absorbed, and it does not enter bodies of water. On high latitudes most fallout occurs when snow covers the earth. In spring when the snow melts, the surface of soil is still frozen, and consequently, the flood is able to carry ^{137}Cs into lakes. During the summer small additional quantities of ^{137}Cs can enter water systems mostly with that part of rain that falls directly to the water surface. The physical state of fallout cesium in fresh water is not known, but since 70% of fallout cesium in sea water is in ionic form (Greendale and Ballou, 1954), it is probable that about the same percentage of fallout ^{137}Cs is in ionic form in fresh water too.

Even though most of the high-level radioactive wastes are stored for decaying at fuel element processing plants, low-level wastes are released into

streams that carry them into lakes and seas. In the future nuclear vessels may be an important source of radioactive wastes in the sea. Laboratories utilizing radioisotopes are now another small scale source of ^{137}Cs.

Methods of Uptake

As soon as radiocesium enters a water body, it is involved in the physico-chemical and biological processes. Factors removing ^{137}Cs from water are sorption by allochthonous and autochthonous particles and biological uptake by organisms. Sorption may occur by adsorption and absorption, both of which may be classified as a specific or nonspecific electrostatic type (Daniels, 1948). Nonspecific electrostatic adsorption occurs mainly in the region of the ionic double layer by electrostatic attractive forces. This is also called "physical adsorption." Specific adsorption is a result of a strong chemical or geometrical interaction between the adsorbed ion and the surface of an adsorbing lattice. Specific adsorption is also termed "chemical adsorption."

Physical sorption is believed to be the predominant method of sorption of isotopes by the coarser grained soils such as silts, sands and gravels (Carroll, 1959). A special type of chemical sorption is ion exchange in which ions in the solution replace ions held by the sorbent (Daniels, 1948). Ion exchange is the most important method of sorption of ^{137}Cs by the finer grained soils such as clays (Carroll, 1959). Ion exchange reactions follow the law of mass-action (Halfferich, 1962) and are stoichiometric and reversible. The reaction between cesium and a clay mineral containing a monovalent exchangeable cation (M) may be expressed (Shih and Gloyna, 1966):

$$Cs^{+} + M \cdot Clay \rightleftarrows Cs \cdot Clay + M^{+} .$$

Illite is the most effective clay mineral to sorb radiocesium (Tamura and Jacobs, 1960; Garder and Skulberg, 1964).

Suspended clay particles are the main factor removing ^{137}Cs from water in turbid eutrophic water bodies, but the uptake by vegetation also plays a significant role (Kolehmainen et al. 1968a; Romantschuk et al. 1968). The removal rate of ^{137}Cs from water in a turbid lake follows an exponential curve with two components. A rapid initial loss is caused by sorption on clay particles and by the uptake of primary producers and the slow component is probably the result of transport of ^{137}Cs with sedimenting clay particles into bottom sediments. The removal rate of ^{137}Cs from water by nonbiotic factors is slow in the bodies of water with no clay turbidity (Kolehmainen et al. 1968a).

Uptake of ^{137}Cs by primary producers involves several mechanisms, viz , nonmetabolic adsorption (physical adsorption), nonmetabolic absorption and metabolic absorption (Gutknecht, 1965a). In algae the nonmetabolic uptake is of greater importance than in higher aquatic plants (Williams, 1960; Gutknecht, 1965a) while in higher aquatic plants uptake by roots is probably an additional method.

The metabolism of cesium is related to the metabolism of potassium and rubidium in such a way that these elements can substitute for each other to some extent (Relman, 1956). The uptake, concentration factor and the excretion of these elements, however, differ from each other in the red alga, Gracilaria

foliifera, which supports the fact that cesium is taken up independently, apparently by active processes (Gutknecht, 1965b).

It is not known whether cesium is necessary for organisms. Although Rice (1963) mentions that cesium is not necessary for plant growth, it is taken up by algae in proportion to the concentration of cesium in water. The concentration factor of cesium stayed the same in algae even though the concentration of cesium in water varied from 0.1 to 13,000 µg/ℓ (King, 1964; Gutknecht, 1965a, b). This means that if the potassium concentration in water is kept constant, primary producers concentrate cesium in proportion to the concentration of cesium in water.

There are two facts that support the opinion that cesium might have a biological function in aquatic plants; (1) an active uptake mechanism by cells, independent from other alkali metals, and (2) the concentration of cesium in aquatic plants in proportion to the concentration of cesium in their medium.

Zooplankton takes little of its radiocesium directly from water, but the main portion of it is absorbed from food, viz, from phytoplankton (Williams and Pickering, 1961; King, 1964). In contrast Gallegos and Whicker (1969) reported the uptake of radioecesium in Daphnia sp. was directly from water rather than from food. The assimilation of ^{137}Cs from food by zooplankton is low, 21% according to King (1964).

Bottom animals assimilate radiocesium through their food too. The assimilation percentage among detritus feeders in lakes turbid with clay particles is probably lower than among bottom animals feeding on plankton, periphyton or higher plants because ^{137}Cs sorbed by clay particles is not as readily

available for organisms as ^{137}Cs in cells. However, specific data are lacking for this step in food chain.

Fresh water fish obtain about 99% of their daily intake of ^{137}Cs from their food and less than 1% is taken directly from water by gills (Garder and Skulberg, 1966; Kolehmainen et al. 1967). The assimilation percentage varies from 7% to 80% depending on the type of the food (Kevern, 1966).

Factors Affecting the Accumulation of ^{137}Cs in Aquatic Organisms

The concentration of radiocesium in phytoplankton, periphyton and higher aquatic plants is regulated mainly by the concentration of radiocesium and potassium in water. One might expect that the concentration of stable cesium would also have a great effect on the concentration factors of radiocesium as occurs with the stable isotopes of other radionuclides (Kevern et al. 1964b; Gutknecht, 1965a), but apparently this is not the case with radiocesium and stable cesium. King (1964) reported that stable cesium in concentrations between 0.1 - 1000 µg/ℓ had no effect on the C.F.'s of ^{137}Cs in Chlamydomonas moewusii. Gutknecht (1965a, b) found the same thing in the marine red alga, Gracilaria foliifera and also in other algae: Fucus sp., Ulva sp. and Porphyra sp. in cesium concentrations of 0.01 - 100 µeq/ℓ (1.33 - 13,300 µg/ℓ). This means that the cesium pool (stable + radioactive isotopes) is concentrated by algae proportionally to the external concentration, and that stable cesium does not have any carrier effect in the concentrations at which it exists in aquatic environments (see page 3). Williams and Swanson (1958) showed that when stable cesium was used in higher

concentrations, > 10,000 times of natural water, it decreased the concentration factor of radiocesium in Euglena intermedia and that in those concentrations stable cesium acted as a diluent for radiocesium.

The role of potassium as a nonisotopic carrier for ^{137}Cs in primary producers has been shown both in laboratory and field conditions. In laboratory experiments an increase in the extracellular potassium concentration decreased the C.F. of ^{137}Cs in algae (Williams and Swanson, 1958). Sabo and Bedrosian (1963) found the C.F. of ^{137}Cs to be 3.1 in the marine alga Nannochloris while Williams and Swanson (1958) reported the C.F. of ^{137}Cs in fresh water algae to vary from 52 to 1530. Sabo and Bedrosian (1963) attributed the low concentration factor of ^{137}Cs in Nannochloris to the high concentration of potassium in sea water. The inverse relationship between the concentration of radiocesium in higher aquatic plants and the concentration of potassium in water has been shown in laboratory experiments (Beninson et al. 1966) and also in lakes (Kolehmainen et al. 1967, 1968a, b, c).

The role of rubidium as a carrier for ^{137}Cs in aquatic ecosystems is unknown. In rats the retention of ^{137}Cs is affected less by the concentration of rubidium in food than by the concentration of potassium in food (Johnson et al. 1968). Rubidium also proved to be toxic even when fed in concentrations of 0.1% in food.

In algae the size of the algal cell affects the C.F. of many radioisotopes that are concentrated partly by surface adsorption (Rice, 1963). Cesium-137 is accumulated partly by surface adsorption, and the content of ^{137}Cs in algae

should be higher in species that have smaller cells than in the species that have larger cells, because the smaller the cell the greater the surface area per unit of biomass. Williams (1960) found that most of radiocesium in the alga Chlorella was bound to cell wall which also supports the effects of the cell size.

The physiological age of the specimens also affects the C.F. of the species and the effect seems to be different in different species. Williams (1960) found that physiologically old cells concentrated more ^{137}Cs than physiologically young cells in the green alga Chlorella sp. while Pendleton (1959) reported just the opposite case in filamentous green algae, Rhizoclonium crassipellitum and Spirogyra crassa.

Algae accumulate more ^{137}Cs in the light than in the dark (Williams, 1960; Gutknecht, 1965a), and the rates of uptake and elimination of potassium, rubidium and cesium in algae are faster in the light than in the dark (Gutknecht, 1965b).

Seasonal fluctuations of ^{137}Cs concentration in organisms in radioactive effluents, probably caused by the fluctuation of temperature, were reported by Pendleton (1959). Seasonal fluctuation of fallout ^{137}Cs concentration in organisms has a lag-time from the seasonal fluctuation of the ^{137}Cs concentration in water, the length of which seems to depend on the length of the food chain and the biological half-life of ^{137}Cs in the organism.

All animals accumulate ^{137}Cs mainly or exclusively from their food. Thus, the concentration of ^{137}Cs in aquatic animals depends on the concentration of ^{137}Cs in their food organisms, on the proportion of ^{137}Cs assimilated from food, and on the rate at which it is excreted or its biological half-life. Pendleton et al. (1965) suggested that the concentration of ^{137}Cs

increased with the length of food chain. This suggestion is called the "Trophic level effect." The effect is based on the fact that the biological half-life of cesium is 2 to 3 times longer than that of potassium and that cesium is assimilated as efficiently as potassium in metabolic processes. This "Trophic level effect," however, is not true on all food chains, both in terrestrial ecosystems (Reichle and Crossley, 1965, 1969) or in aquatic ecosystems (Williams and Pickering, 1961; King, 1964; Nelson, 1969; Kolehmainen et al. 1968a). One reason this theory does not hold true is that the assimilation of ^{137}Cs varies greatly depending on the type of food and the species in question.

The concentration of ^{137}Cs in zooplankton is mostly lower than that of algae (Kolehmainen et al. 1968a), probably caused by a low assimilation ratio and short biological half-life. The concentration of radiocesium in bottom animals is at the same level as that of zooplankton, so there is not an enrichment from the primary producers up to bottom animals (Kolehmainen et al. 1968a, c). Fish, however, have higher concentrations than organisms on lower trophic levels, and there is even an increase in concentration of ^{137}Cs from small fish to predatory fish (Häsänen and Miettinen, 1963; Kolehmainen et al. 1964, 1966, 1967, 1968c; Pendleton, 1965; Gustafson et al. 1966; Hannerz, 1966, 1968). Predatory fish usually have 2 to 3 times as much ^{137}Cs as fish feeding on plankton and bottom animals.

The size of fish affects the concentration of radiocesium in at least two ways: (1) small fish have a shorter biological half-life of ^{137}Cs (Häsänen et al. 1967, 1968), and (2) many species of fish change their feeding habits when they

grow; small fish consume zooplankton or bottom animals, and as they grow, become predators on small fish. Hannerz (1966) found an increase of 2.6 times in the concentration of ^{137}Cs in the flesh of perch, Perca fluviatilis, with an increase of 4 cm to 26 cm, and an increase of two times in the flesh of burbot, Lota vulgaris, as fish grew from 33 to 60 cm length. Large perch (> 20 cm) in Finnish lakes usually have 2 to 2.5 times as high concentrations of ^{137}Cs as small perch (< 20 cm) in the same lake (Kolehmainen et al. 1968b, c) and this difference is caused by the changing of feeding habits from bottom animals to small fish. The percentage of ^{137}Cs assimilated from small fish might also be higher than the percentage of assimilation from bottom animals.

Comparison of the ^{137}Cs Concentrations in Organisms in Different Types of Lakes

Ecological factors are responsible for some differences in ^{137}Cs concentrations between different trophic levels. Zooplankton in oligotrophic lakes usually has a higher ^{137}Cs content than bottom animals while in eutrophic lakes the situation is reversed. In eutrophic lakes ^{137}Cs is deposited into bottom sediments and vegetation, and is more readily available for bottom animals than for zooplankton. The rate of sedimentation is slow in oligotrophic lakes and the sediments do not accumulate ^{137}Cs as effectively as in eutrophic lakes; consequently, ^{137}Cs in oligotrophic lakes is more easily available for zooplankton than for bottom animals (Kolehmainen et al. 1968a).

In Finnish lakes ^{137}Cs seems to be concentrated according to the length of food chain from the level of primary consumers up to fish. The concentration

of ^{137}Cs increases along both types of food chains: zooplankton-fish-predatory fish, and bottom animals-fish-predatory fish (Kolehmainen et al. 1968c). In highly turbid, eutrophic lakes the concentration processes in the food chains do not show the same "Trophic level effect" (see p. 16), but each trophic level seems to behave differently (Kevern and Griffith, 1966; Nelson, 1969). The reason for this kind of discrepancy is not known.

Factors that contribute to a slow removal of ^{137}Cs in oligotrophic lakes are (1) no clay turbidity and (2) low primary production. Because of the slow removal rate, fallout ^{137}Cs is accumulated in oligotrophic lakes, and the highest concentrations of ^{137}Cs in lake water were obtained in summer 1964, one year after the highest peak of fallout (see Bortoli et al. 1964, 1965a,b, 1966, 1967a; Salo, 1966, 1967).

Even though the maximum of fallout comes at middle latitudes (Lavrenchik, 1965), the highest concentrations of ^{137}Cs in aquatic organisms have been obtained in lakes at high latitudes (Häsänen and Miettinen, 1963; Gustafson, 1966, 1967; Kolehmainen et al. 1964, 1966, 1967, 1968b,c) because the high latitude lakes are mostly oligotrophic. The highest value reported so far has been 59 nCi/kg fresh weight in perch of a small oligotrophic seepage lake in Finland in 1966 (Kolehmainen et al. 1968b). High values have also been reported in fish from oligotrophic lakes in mountain areas at middle latitudes (Bortoli et al. 1965a,b,

1966, 1967a; Whicker, 1968; Nelson and Whicker, 1969). The annual fallout has been about seven times higher in northern Italy than in Finland (Bortoli et al. 1965a,b, 1966, 1967a; Salo, 1966, 1967). The fish in northern Italian lakes did not, however, reach as high concentrations of ^{137}Cs as the fish in Finnish lakes. The main reason for the differences in the ^{137}Cs level in fishes was that the concentration of potassium in Finnish lakes was lower (Kolehmainen et al. 1966) than in northern Italian lakes (Bortoli et al. 1965b, 1969).

Equilibrium Concept of ^{137}Cs in Aquatic Ecosystems

The fast removal of ^{137}Cs in eutrophic lakes makes it impossible for the higher trophic levels to attain equilibrium with the concentration of ^{137}Cs in water. The time required for a species to reach equilibrium is determined by the length of the food chain and the biological half-life of the species. Phytoplankton, periphyton, and higher plants are probably close to equilibrium, even though the concentration of ^{137}Cs in water varies, because primary producers reach equilibrium in two to four weeks (Williams, 1960; Gutknecht, 1965a; Kolehmainen et al. 1968a). Zooplankton attains equilibrium in about three to four weeks (Hannerz, 1966; Kolehmainen et al. 1968a) and bottom animals in about five weeks (Kolehmainen et al. 1968a; Romantschuk et al. 1968). For fish it takes one to two years to attain equilibrium (Kolehmainen et al. 1968a).

In lakes contaminated with fallout the fluctuation of the concentration of ^{137}Cs in water has been considerable (Bortoli et al. 1964, 1965a,b, 1966, 1967a; Salo, 1966, 1967). This means that animals at higher trophic levels are not at

an equilibrium concentration of ^{137}Cs, but the concentration fluctuates above and below the equilibrium as Bortoli et al. (1967b) showed. Organisms in aquatic ecosystems receiving radioactive effluents may reach an equilibrium or a steady state if the release of radioactive wastes is fairly uniform throughout the years. For this reason rivers and lakes receiving radioactive effluents from nuclear power plants are convenient places to study biological factors involved in the behavior of a radioisotope in an aquatic ecosystem.

CHAPTER III

ECOLOGY OF BLUEGILL IN WHITE OAK LAKE

White Oak Lake was impounded in 1943 to serve as the final control of radioisotopes released from Oak Ridge National Laboratory to the Clinch River. The surface area of this reservoir was 18 hectares at that time. In 1955 the lake was drained and the surface area decreased to 2.8 hectares. During 1955-63 the former lake bottom was covered by thick grass, herb and shrub vegetation. In 1960 the dam was closed and the surface level raised 3 in. which increased the area to 3.2 hectares. In 1963 when Melton Hill Dam was completed White Oak Lake had a surface area of 6 hectares and presently the lake covers an area of 10.5 hectares. White Oak Lake is fairly shallow with a maximum depth of 2.4 m. The mean annual temperature in the whole water column during the study was 15.8 C, with extremes of 29.5 and 3.0 C.

The water of White Oak Lake is turbid with clay particles and the bottom sediments contain much clay material. Since 1943 large quantities of ^{106}Ru, ^{137}Cs, ^{60}Co and ^{65}Zn have been deposited into the bottom sediments (Lomenick and Gardiner, 1965).

White Oak Lake does not have higher aquatic plants, but phytoplankton production is high and benthic bottom fauna is well developed. Most common forms are Diptera larvae belonging to the families Chironomidae, Ceratopogonidae and Chaoboridae. The densities of the fish populations were high after the surface level was raised in 1963, but during the time of this research the populations

have been small. The productivity and the species composition in White Oak Lake were quite different in 1967-1968 from those during the period from 1943 to 1955 (Krumholz, 1956).

The composition of fish species has been changing during the period beginning in 1963. Three species; carp, smallmouth buffalo, and black bullhead have probably disappeared by now. The fish species in White Oak Lake presently are:

bluegill, Lepomis macrochirus (Rafinesque)

redear sunfish, Lepomis microlophus (Gunther)

warmouth, Chaenobryttus gulosus (Cuvier)

largemouth bass, Micropterus salmoides (Lacépède)

gizzard shad, Dorosoma cepedianum, (LeSueur)

golden shiner, Notemigonus crysoleucas (Mitchill)

goldfish, Carassius auratus (Linnaeus)

mosquitofish, Gambusia affinis (Baird and Girard)

Fish were sampled by hoop nets, gill nets, electrical shocker, seining and pole. In spite of the variety of sampling methods a sufficient number of fish could not be sampled every time. The total number of bluegill collected was 285. Large bluegill were sampled most effectively because they were easier to catch and because size did not affect metabolism in larger fish as much as in small fish. The average weight of bluegill sampled frequently was 107.6 ± 43.8 g. Some samples of small bluegill were also taken for comparison. Fish were brought to the laboratory alive and stored in plastic bags in a

freezer if not prepared immediately for analysis. Fish samples were taken at least twice a month, if possible, during September 1967 - October 1968. Some fish were collected already in June 1967. A sample usually consisted of five to ten fish. Besides bluegill other species of fish were collected for comparison.

The growth of bluegill was determined using the method of the total length-scale relationship (Carlander and Smith, 1944). Total length of each fish was measured in millimeters. The scales were taken from the left side of the body near the tip of the pectoral fin when it is pressed to the body parallel to the main axis (Regier, 1962).

Five scales from 74 fish were used for age and growth determination. Scales were washed, pressed on cellulose acetate sheets with a Carver laboratory press, and the radii of the annulii were measured with an Eberbach scale reader. The equation for total length-scale relationship calculated by the least squares method was $Y = 11.09 + 47.33\ X$, where Y was the total length in mm, and X was the length of the anterior radius of the scale in mm. The total length of each age group is given in Table 3. The growth of the White Oak Lake bluegills was similar to the growth of bluegill of other lakes in midwestern and southcentral states (Lane, 1954).

The length - weight relationship in females followed the equation $Y = -5.169 + 3.227\ X$, where Y is the logarithm of the weight (g) and X the logarithm of the total length (mm). The equation for males was $Y = -5.712 + 3.469\ X$. The weight of each age group is given in Table 3. The length of

TABLE 3

CALCULATED AVERAGE TOTAL LENGTHS (mm) AND AVERAGE WEIGHTS (g) FOR EACH AGE-GROUP OF MALE AND FEMALE BLUEGILL

	Age-Group						No. of Fish
	I	II	III	IV	V	VI	
Total Length:							
Males	46	120	151	164	173	180	31
Females	43	115	145	161	171	179	43
Weight:							
Males	1.1	31.7	68.7	93.6	112.3	129.3	31
Females	1.0	30.3	64.9	89.6	108.7	128.7	43

growing season was determined from the time of annulus formation using the method described by Gerking (1966). The formation of the annulus in White Oak Lake bluegill began in early April and this was considered as the beginning of the growing season. Ninety percent of the annual growth was achieved by the middle of November. Thus, the growing season was considered to be seven and a half months although some growth occurred during the remainder of the year.

The feeding habits of bluegill were studied by examining the stomach contents of each fish. Since fish were collected partly by hoop nets that were emptied every two or three days, the fullness of stomach could not be used as an indication of feeding activity. Most of the fish had some food in their stomachs and these items were sorted and identified.

The volume of each main food item in the stomach contents was estimated under a microscope. The validity of the visual estimation was checked by weighing some stomach samples after they had been separated into groups. The agreement between visual and gravimetric estimates was good and the volumetric estimation was applied throughout the remaining samples. Low population density of the bluegill made it impossible to collect fish frequently enough during a 24 hours period to determine the daily quantitative food consumption on the basis of the stomach fullness.

In June some individuals had eaten entirely roe, and in August terrestrial insects, but for the most part of the year the diet consisted of Chironomid larvae, other insect larvae, plants, and detritus in the proportions given in Table 4. The feeding habits of bluegill were similar to those in other lakes (Gerking,

TABLE 4

VOLUME OF THE MAIN FOOD ITEMS IN THE STOMACH CONTENTS OF 153 BLUEGILL

Food Item	Percentage
Chironomid Larvae	56.1
Chironomid Pupae and Adults	6.4
Other Insects: Larvae, Pupae and Adults	14.4
Roe	3.5
Fish	0.9
Plant Material	4.9
Detritus and Sediments	9.6
Others	4.2
	100.0

1962; Keast and Welsh, 1968) except that no cladocera were found in the stomachs of bluegill in White Oak Lake. The food of bluegill in White Oak Lake consisted of 43% plant remains and algae, and only 11.5% Chironomid larvae in 1952 (Krumholz, 1956).

CHAPTER IV

CESIUM-137, STABLE CESIUM AND POTASSIUM BALANCES IN THE WHITE OAK LAKE BLUEGILL

Introduction to the Radioisotope Method in Studying the Feeding Rates in Fish

The balance of a radioisotope in fish can be simulated with a one-compartmental system where the uptake (input) and excretion (output) determine the body burden. If the radioisotope is taken from food by fish, the balance of the radioisotope can be applied to calculate the food consumption rate in fish. When the body burden and the excretion rate of the isotope in fish are determined, the intake of the isotope can be calculated. If the concentration of the isotope in food and the assimilation percentage of the isotope for food are known, it is easy to calculate the quantity of food consumed daily by fish. While calculating these it is necessary to know whether fish are in equilibrium with the isotope because this determines the type of the equations that can be applied.

The advantage of radioisotopes in studying the food intake in fish was first recognized by Davis and Foster (1958). They calculated the daily intake of radiophosphorus in an equilibrium situation using the formula

$$Q_e = \frac{ar}{k} \qquad (4.1)$$

where (r) is the daily intake of the radioisotope in food, (a) the proportion of the radioisotope assimilated from food, (Q_e) the body burden of the radioisotope

and (k) the fraction of body burden excreted per day. When the concentration of the radioisotope in food, (d), is known, the quantity of the daily meal, (r'), can be calculated:

$$r' = \frac{r}{d} \quad . \tag{4.2}$$

In the equilibrium situation the daily food intake can be calculated directly if the equation (4.2) is substituted in the equation (4.1):

$$r' = \frac{Q_e k}{ad} \quad . \tag{4.3}$$

Kevern (1966) modified this equation to a situation where fish fed on several food items that had different concentrations of the radioisotope and different assimilation percentages:

$$r' = \frac{Q_e k}{\sum_{i=1}^{n} a_i d_i f_i} \quad , \tag{4.4}$$

where (a_i) is the fraction of the radioisotope assimilated from (i)th food item, (d_i) the concentration of the radioisotope in (i)th food item and (f_i) the fraction of (i)th item in the diet.

If the concentration of a radioisotope in food or the food consumption varies or both vary in the course of time, the equations based on the equilibrium state cannot be applied to calculate the balance of the isotope or the consumption of the food. In a fallout situation both the concentration of ^{137}Cs in fish and the concentration of ^{137}Cs in the food of fish change continuously. The following equation was used to calculate the intake of ^{137}Cs and the daily food consumption on the basis of fallout radiocesium in fish (Kolehmainen et al. 1967):

$$A_t = A_o\, e^{-kt} + \frac{ar}{k}(1-e^{-kt}) \quad , \tag{4.5}$$

where (A_t) was the concentration of the radioisotope in fish at time (t), (A_o) the concentration of the radioisotope in fish initially, and (a), (r), and (k) as defined earlier. By deriving this equation for the daily intake of the radioisotope (I) or (ar) it gives:

$$I = a \cdot r = \frac{k(A_t - A_o e^{-kt})}{1 - e^{-kt}} \quad . \tag{4.6}$$

If the equation (4.2) is substituted into this, it gives:

$$r' = \frac{k\,(A_t - A_o e^{-kt})}{ad\,(1-e^{-kt})} \quad . \tag{4.7}$$

Cesium-137 Balance in Bluegill

The balance of radiocesium in the bluegill was studied by (1) analyzing the body burden of radiocesium during the period of 20 months, (2) determining the biological half-life of cesium, and (3) studying the factors involved in the uptake of ^{137}Cs from food, including the feeding habits of bluegill in White Oak Lake, the concentration of ^{137}Cs in different food items, and the proportion of ^{137}Cs assimilated from different food items.

Body Burden of ^{137}Cs in Bluegill

In the beginning of the study, five to ten fish were composited and counted as one sample. The flesh was analyzed separately from the rest of the bodies, and the entrails were excluded from the samples. Afterwards it was

found that a great difference existed in the concentration of ^{137}Cs between different individuals, and thereafter the fish were counted individually. For modeling purposes it was better to include the entrails, after the removal of food remains, into the samples to get the total body burden of ^{137}Cs. The removal of the entrails lowered the concentration of ^{137}Cs in fish 9.5% on the average. The values of ^{137}Cs in the samples not containing entrails were corrected with the percentage above. Fish samples were analyzed γ-spectrometrically with a 400-channel Packard, or with a 1024-channel Nuclear Data pulse height analyzer in conjunction with a dual crystal assembly (NaI (Tl), 3" x 3"). Besides ^{137}Cs the samples contained ^{106}Ru, ^{60}Co and ^{65}Zn and for this reason the quantity of each radionuclide was calculated by a digital computer with a program that calculated the individual radionuclides by a linear least squares method. The concentrations of ^{60}Co was 10 to 15% of that of ^{137}Cs and the concentrations of ^{106}Ru and ^{65}Zn were even less. The results of the ^{60}Co, ^{106}Ru and ^{65}Zn measurements are not discussed.

The concentration of ^{137}Cs in the bluegill showed three types of variations: (1) the concentration increased with the weight of the fish, (2) the concentration followed a seasonal trend with a maximum in February and a minimum in August, and (3) the individual variation in the same size of fish taken on the same day was up to five times.

The increase of the concentration of ^{137}Cs with the weight was linear up to 70 grams ($Y = 9.26 + 0.390\ X$, $r^2 = 0.998$), where (Y) was the concentration of ^{137}Cs in fish per gram and (X) the weight of fish in grams. No correlation existed between the concentration of ^{137}Cs and the size of the fish in fish above

70 grams, which probably means that the concentration of ^{137}Cs in fish above 70 grams was in a steady state. The average concentration of ^{137}Cs in bluegill of different sizes are given in Table 5. The concentration of radiocesium in bluegill above 70 grams at different times during the study is shown in Fig. 2. The sinusoidal curve was fitted to the data by a least squares analysis. The equation for this sine curve was:

$$Y = \sin \left[\frac{2\pi}{360} \cdot (t - 313)\right] \cdot 9.2 + 38.2 \qquad (4.8)$$

where (Y) is the concentration of ^{137}Cs (pCi/g fresh weight) and (t) the number of the day in the year. The highest individual value was 151.20 and the lowest 19.67 pCi/g in fish over 70 grams. The average concentration in fish above 70 grams was the same as Kevern and Griffith (1966) reported in the White Oak Lake bluegill in January 1966.

The concentration of dissolved ^{137}Cs in the water of White Oak Lake is also shown in Fig. 2. These values were calculated on the basis of the quantity of ^{137}Cs in total water samples analyzed by the Environs Monitoring Section of Health Physics Division at ORNL (courtesy W. D. Cottrell). Lammers (1968) found that 61.5% of ^{137}Cs obtained in water samples from White Oak Lake was in the particulate fraction hence 38.5% was assumed to be dissolved. In five water samples taken during the last part of 1968 the proportion of ^{137}Cs in the particulate fraction was 59.9%. This was very close to what Lammers (1968) determined. However, Lammers' value was used for the calculations because he had a more exact centrifuging procedure and also more samples. The mean concentration of

TABLE 5

CONCENTRATION OF ^{137}CS IN BLUEGILL OF DIFFERENT SIZES IN WHITE OAK LAKE

Weight of Fish g		Concentration of ^{137}Cs pCi/g		No. of Fish
Mean	SD	Mean	SD	
2.1	1.8	10.43	5.59	31
10.5	6.8	13.28	4.32	13
32.0	6.4	21.21	12.69	19
55.1	7.1	30.90	10.52	13
78.0	5.1	39.79	12.69	20
109.6	32.1	40.12	15.21	186

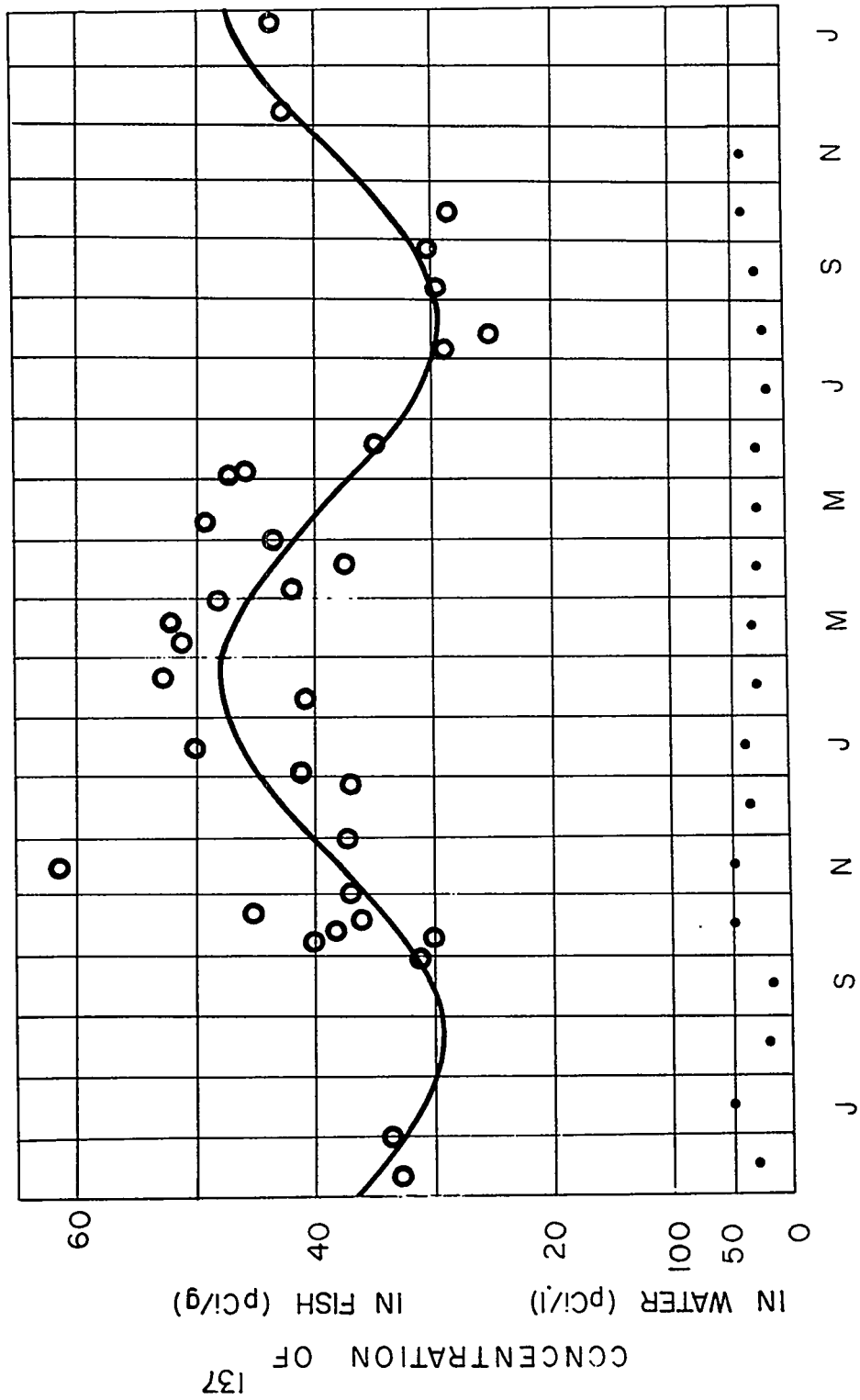

Figure 2. Seasonal cycling of ^{137}Cs concentration in bluegill (> 70 g) and the concentration of dissolved ^{137}Cs in White Oak Lake water.

^{137}Cs in total water samples was 150.5 pCi/ℓ and the concentration of dissolved ^{137}Cs 58.2 pCi/ℓ during 1967-68. Because of the seasonal trend and the great individual variation of ^{137}Cs concentration in fish, it was impossible to see any correlation between the change in the concentration of radiocesium in water and in fish.

Biological Half-life of ^{137}Cs in Bluegill

The T_b of ^{137}Cs in fish consists of two components (Kevern et al. 1964a; Kevern, 1966; Häsänen et al. 1967, 1968). The length of T_b varies among different species of fish (Häsänen et al. 1967, 1968), and in the same species the length of T_b is a function of the temperature (Kevern et al. 1964a; Kevern, 1966, Häsänen et al. 1967, 1968) as well as a function of the size of fish (Morgan 1964; Häsänen et al. 1967, 1968). The effect of temperature on T_b was not studied in bluegill but it was assumed to follow the Q_{10} factor ($Q_{10} = 2$):

$$T_{b_t} = T_{b_o} \cdot e^{0.0693\ (\phi_o - \phi_t)}\ , \qquad (4.9)$$

where (T_{b_t}) is the biological half-life at temperature (ϕ_t) and (T_{b_o}) the biological half-life at temperature (ϕ_o). This means that the length of T_b doubles as the temperature of water is lowered 10 C.

All the previous studies on T_b of cesium in fish have been performed with fish that have been "tagged" either with a single feeding of radiocesium or with a short period of feeding "tagged" food or with keeping the fish for a short time in water containing radiocesium. This means that the fish have not been near equilibrium state when T_b has been determined. To see whether any

differences exist in T_b between fish that are in equilibrium with radiocesium in the environment and fish that receive a single feeding of radiocesium, T_b experiments were carried through with fish taken from White Oak Lake as well as with fish that were fed a known quantity of ^{137}Cs.

All experiments were conducted in the laboratory except one that was made outdoors in a concrete pond (6.5 cubic meters). Large bluegill were kept in 400 ℓ aquaria ("Living Streams," Frigid Units Co.) in a continuous flow and small bluegill and fingerlings in 2 to 10 ℓ plastic containers.

Fish taken from White Oak Lake had a small body burden of ^{137}Cs; and since they also had a small quantity of other radioisotopes, the retention of ^{137}Cs was counted with the Packard Model 15 or with the Nuclear Data 2200 multichannel analyzer in conjunction with the dual crystal assembly. The fish were placed for counting in a plastic bag in a bluegill-shaped plastic box. Fish that were tagged by force-feeding a known quantity of ^{137}Cs in a gelatin capsule were counted with a Packard Auto-Gamma Model 410 single-channel analyzer in conjunction with an Armac scintillation detector using the above mentioned counting box to provide a constant counting geometry. Small bluegill were counted with Packard 410 single-channel analyzer placed in a plastic bag in a fish shaped plastic box, and bluegill fingerlings (20 specimens) were counted as one group in a plastic counting dish containing 80 ml water. Large bluegill were anaesthetized with Tricaine Methane-sulphonate (MS 222 - Sandoz) while tagging and counting to make the handling less stressing for the fish. Bluegill seemed to be very sensitive to any kind of manipulation, and they became

frequently exposed to fungus, Saprolegnia sp., especially in water temperatures above 18 C. The infection of fungus, however, did not affect the results because the infected specimens were removed as soon as the infection became apparent. Large bluegill were fed with commercial dry food, earth worms, and Chironomus larvae while small bluegill and bluegill fingerlings were fed entirely with Chironomus larvae.

The excretion of ^{137}Cs after a single feeding was studied with a group of 12 bluegill (80 - 120 g) that were force-fed a small gelatin capsule with 125 nCi of ^{137}Cs solution. The fish were kept in a "Living Stream" unit in a continuous flow at 15.5 ± 0.7 C (one standard deviation, SD). The mortality in this experiment was high because fish had to be counted often. The experiment had to be repeated twice before the excretion could be followed long enough to determine the long component of T_b. The excretion curve was identical in all the groups as long as the experiment could be continued, thus the results were reliable despite the heavy mortality.

The length of the short component (T_{b_1}) was 7.8 ± 0.5 (SD) days and of long component (T_{b_2}) 183.0 ± 11.0 (SD) days, and the proportions of the assimilated tracer that were excreted at each rate 36.9 ± 4.0 (SD)% and 63.1 ± 4.0 (SD)%, respectively. The values were analyzed for each individual separately by calculating first T_{b_2} by a least squares analysis on the basis of the linear part of the retention curve on a semilog plot (Fig. 3). T_{b_1} was calculated the same way after subtracting the values of T_{b_2} from the corresponding data points (Richmond, 1958). After "peeling off" T_{b_1} a third component (T_{b_o}) could be

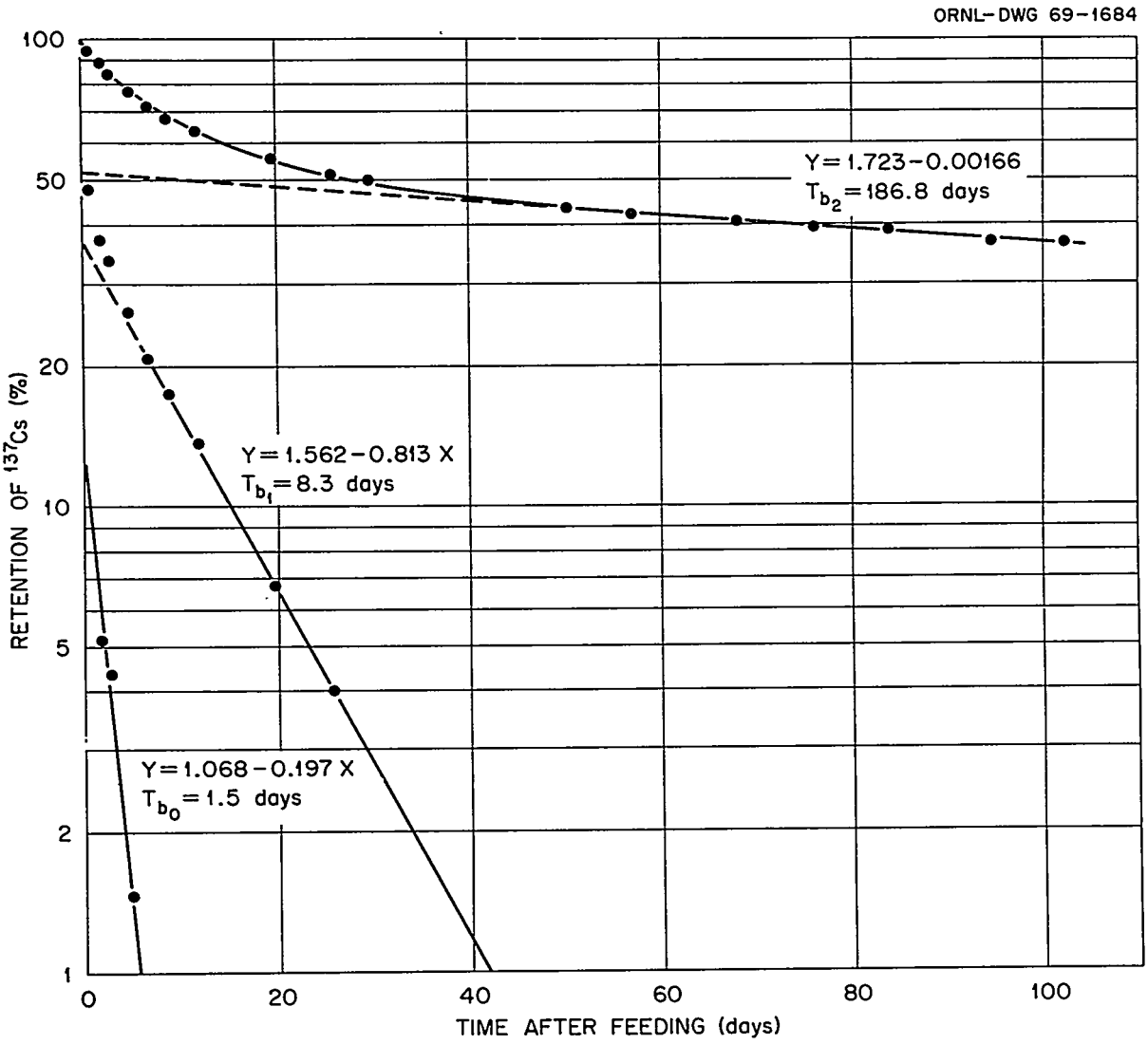

Figure 3. Retention and T_b of ^{137}Cs in a bluegill (72 g) at 15.5 C after a single feeding of ^{137}Cs.

This individual had the best fit.

seen. This represented the passing of the unassimilated quantity of ^{137}Cs through the gastrointestinal tract. The assimilated proportion of ^{137}Cs was 91.6 ± 3.6 (SD)% in this experiment.

The biological half-life of ^{137}Cs in the White Oak Lake bluegill was determined with two groups of fish that were brought from the lake for determining the retention of ^{137}Cs. One group of 12 fish (60 - 80 g) was kept in a "Living Stream" unit at 14.5 ± 0.6 (SD) C, and the other group of eight fish (80 - 100 g) was kept in a concrete pond outside the building at 15.6 ± 2.7 (SD) C.

The excretion of ^{137}Cs followed a similar pattern in both of these groups. The length of T_{b_2} was 219.8 ± 53.4 (SD) days in the group in the laboratory and 188.8 ± 58.2 (SD) days in the group in the concrete pond. When the values were corrected to the mean temperature of White Oak Lake (15.8 C) they were 194.2 ± 47.3 (SD) days and 186.7 ± 58.2 (SD) days, respectively. Since there was no significant difference in T_{b_2} ($F = 0.12$) between the group that was kept in the laboratory and in the group that was kept outdoors in the concrete pond under more natural environmental conditions, the values of T_b obtained in the laboratory were considered reliable for the conditions in White Oak Lake.

The retention curve of ^{137}Cs in the White Oak Lake bluegill was quite different from the retention curve of the bluegill that received a single feeding of ^{137}Cs (Figs. 3 and 4). The T_{b_1} of White Oak Lake fish was indistinguishable as was expected in fish close to an equilibrium state. Since these fish were brought from White Oak Lake in October, they probably were very close to the calculated theoretical equilibrium concentration of ^{137}Cs, 38.2 pCi/g fresh weight (Fig. 2, p. 34).

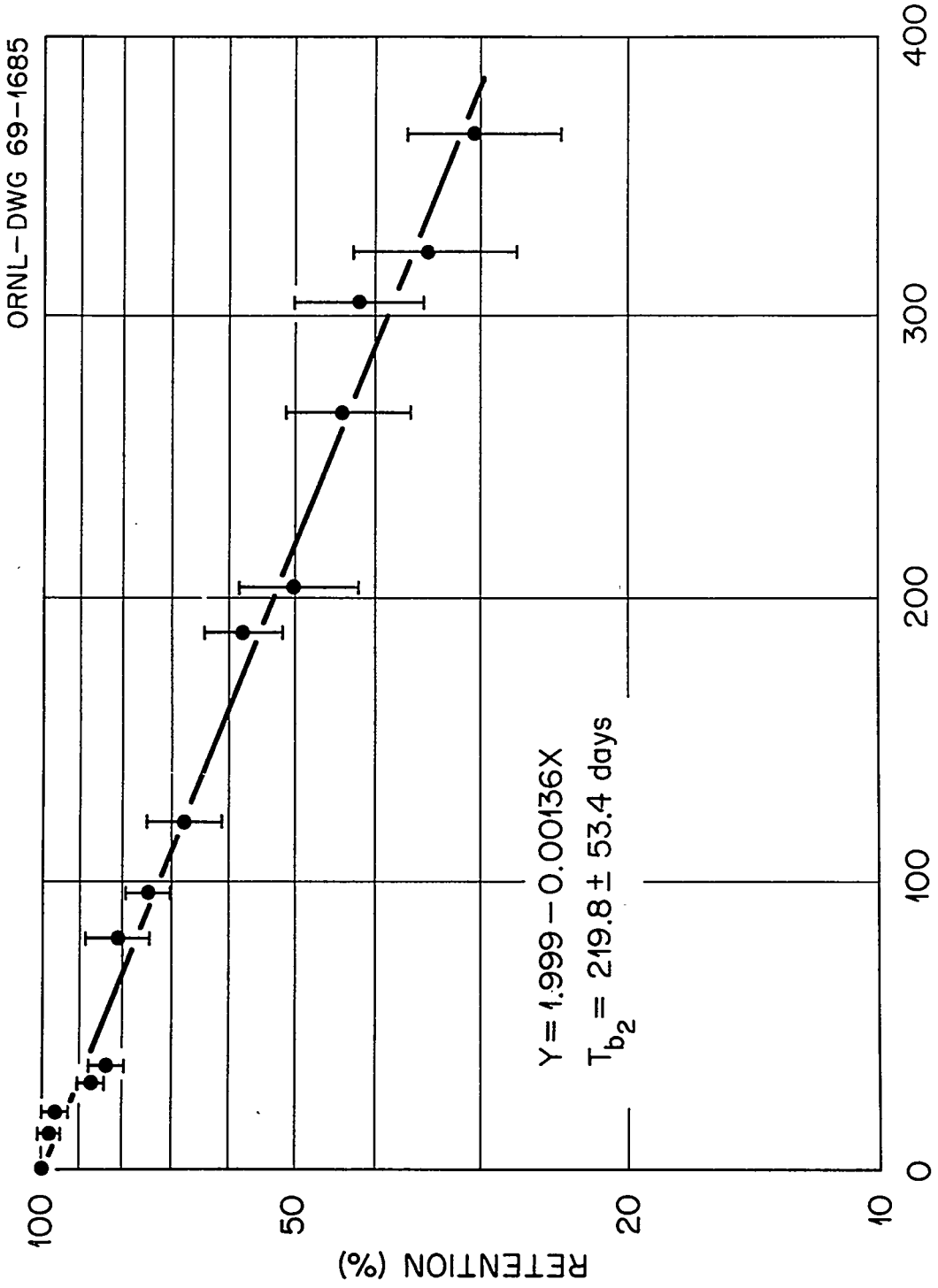

Figure 4. Retention and T_b of ^{137}Cs in White Oak Lake bluegill (n = 12) at 14.5 C.

The proportions of body burden that are excreted by the rate T_{b_1} and T_{b_2} are (a_1) and (a_2), respectively. In an equilibrium state (a_1) can be calculated on the basis of the proportions of ^{137}Cs in the daily intake that are excreted by T_{b_1} and T_{b_2} (p_1 and p_2) using the equilibrium equation (4.1, p. 28). The body burden expressed in terms of assimilated intake (I) and the excretion rate (k) is:

$$Q_e = \frac{I}{k} \quad . \tag{4.10}$$

The excretion rate constant (k) can be calculated on the basis of T_b (expressed in days):

$$k = \frac{\ln 2}{T_b} \quad . \tag{4.11}$$

The body burden can be subdivided into two compartments ($q_1 = a_1 Q_e$) and ($q_2 = a_2 Q_2$) that represent the quantities in the body burden excreted by the rates T_{b_1} and T_{b_2} such that $Q_e = q_1 + q_2$. The excretion rate constants can be calculated separately for T_{b_1}, (k_1), and T_{b_2}, (k_2). The proportions of the daily assimilated intake that go into the two compartments, (q_1) and (q_2), are (p_1) and (p_2) which actually are the probability values for ^{137}Cs atoms for going into either one of the two compartments in this stochastic system. Therefore, $p_1 + p_2 = 1$. Now the equation (4.10) can be written:

$$q_1 = \frac{I \cdot p_1}{k_1}$$

$$q_2 = \frac{I \cdot p_2}{k_2} \quad , \tag{4.12}$$

and since $Q_e = q_1 + q_2$, (4.12) can be combined:

$$Q_e = \frac{I \cdot p_1}{k_1} + \frac{I \cdot p_2}{k_2} = I\left(\frac{p_1}{k_1} + \frac{p_2}{k_2}\right) \quad . \qquad (4.13)$$

Applying the proportions of T_{b_1} and T_{b_2}, and the length of T_{b_1} (corrected for the temperature) determined in bluegill after a single feeding of ^{137}Cs (p. 37), the proportions of the body burden that were excreted by T_{b_1} and T_{b_2} in the White Oak Lake bluegill in the laboratory at 14.5 C were calculated

$$Q_e = I\left(\frac{0.369}{0.0829} + \frac{0.631}{0.00315}\right) = I(4.45 + 200.3). \quad (4.14)$$

The equilibrium value would be 204.8 times the daily intake, and the proportion of the equilibrium body burden excreted by T_{b_1} (a_1) would be:

$$a_1 = \frac{4.45}{204.8} \cdot 100 = 2.17\% \quad . \qquad (4.15)$$

The proportion of the body burden that was excreted by T_{b_2}, (a_2), would be then 97.83%. The linear regression line intercepted the ordinate at 99.8% in this group (Fig. 4, p. 40) and at 100.1% in the group that was kept in the concrete pond, thus the short component could not be recognized in these experiments. However, this 2% error was still inside the statistical error.

There was no significant difference in T_{b_2} between the White Oak Lake bluegill and the bluegill that received a single feeding of ^{137}Cs ($t = 0.254$), thus the T_{b_2} of ^{137}Cs in an equilibrium state was equal to T_{b_2} in a non-equilibrium state. The only difference was in the proportion of the body burden excreted by T_{b_1}, which was the greatest after a single feeding of the radioisotope and decreased gradually into an insignificant proportion when the equilibrium state was achieved.

In an experiment that was designed for determining the assimilation of ^{137}Cs for "tagged" Chironomus larvae in two bluegill (9 and 10 g), T_{b_2} was determined after a 22 days' feeding period. In this experiment fish were kept in small plastic containers in 2 ℓ of water at 12.5 ± 0.5 (SD) C. The long components of T_b were in these fish 135.2 d and 147.0 d or 141.1 d on the average.

Twenty fingerlings were fed once with "tagged" Chironomus larvae and the T_b was determined by following the retention of ^{137}Cs in the whole group. Fish were kept in 5 ℓ of water at 17.5 ± 0.5 (SD) C. The length of T_{b_1} was 5.7 d and T_{b_2} was 76.5 d while the proportions were 20.6% and 79.4%, respectively. The length of T_{b_2} increased slowly with the weight of the fish and the linear regression was $Y = 87.85 + 47.55\ X$ ($r^2 = 0.958$), where Y is T_{b_2} in days and X is the logarithm of the weight of fish. The biological half-lives of ^{137}Cs in different sized bluegill at the annual mean temperature of White Oak Lake (15.8 C) as well as the proportions of T_{b_1} and T_{b_2} are given in Table 6.

Concentration of ^{137}Cs in Bluegill's Food and the Assimilation of ^{137}Cs

The intake of ^{137}Cs was studied by analyzing the concentration of ^{137}Cs in the food of bluegill and by determining the assimilation percentage of ^{137}Cs from different types of food items. The concentration of radiocesium in bluegill's food was studied by counting the stomach contents and food organisms of bluegill sampled from White Oak Lake for ^{137}Cs.

Stomach contents were removed from the stomach, weighed, studied under the microscope and dried at 105 C. After the dry weight was recorded,

TABLE 6

COMBINED RESULTS OF ^{137}CS T_b EXPERIMENTS IN BLUEGILL CALCULATED FOR THE TEMPERATURE 15.8 C

Size of Fish g	T_{b_1} d	p_1 %	T_{b_2} d	p_2 %
0.5 - 1.2	6.4	20.6	86.1	79.4
9 - 10			112.2	
80 - 120	7.6	36.9	187.1	63.1

samples were ashed at 450 C for three days. The temperature had to be kept low to prevent cesium from volatilizing. Ashed samples were treated with concentrated hydrochloric and nitric acid to complete the oxidation, and dissolved in 0.1 N hydrochloric acid.

The samples were analyzed for γ-nuclides with a Packard 400-channel analyzer in conjunction with a 3" x 3" NaI (Tl) well crystal and an automatic sample changer. The quantity of ^{137}Cs was calculated by a digital computer program because the stomach samples also contained some ^{106}Ru and ^{60}Co.

The concentration of ^{137}Cs in the stomach samples followed a seasonal trend with maximum values in the winter and minimum values in the summer (Table 7). The low population density of bluegill unfortunately made it impossible to get stomach samples frequently enough to determine the fluctuation of the radiocesium concentration in bluegill's food every month during the study.

Chironomus larvae and other bottom animals were collected five times during the study by collecting bottom sediments with a long-handled net and the material was sieved through a set of screens. The final picking of bottom animals was done in the laboratory. No attempt was made to get the larvae to empty their gut contents because fish in nature eat them with the material that happens to be in the alimentary system.

Detritus samples were collected by taking bottom sediment samples from the uppermost one-centimeter thick layer and separating detritus from clay minerals by centrifuging the samples at 2000 rpm for 10 minutes. Detritus consisted approximately of the same proportion of clay particles after the centrifuging

TABLE 7

CONCENTRATION OF ^{137}CS IN STOMACH CONTENTS OF BLUEGILL

Date	^{137}Cs in Stomach Contents pCi/g fresh wt	No. of Fish Sampled
Sept. 29, 1967	81.3	11
Nov. 2, 1967	100.4	3
Dec. 1, 1967	105.1	4
Jan. 4, 1968	148.6	4
Feb. 20, 1968	109.4	3
March 11, 1968	74.1	2
April 5, 1968	79.6	14
April 18, 1968	65.7	5
June 4, 1968	68.0	7
June 20, 1968	60.9	4
July 26, 1968	55.6	2
Aug. 6, 1968	58.4	12
Aug. 14, 1968	17.9	3
Sept. 7, 1968	85.1	5
Sept. 27, 1968	87.3	5
	$\bar{x}$ 79.9	Σ 84

as was observed in the detritus in stomach samples (12.6%). The concentrations of ^{137}Cs in the food items of bluegill are given in Table 8.

Since the stomach samples contained some clay, and bottom animals had clay particles in the alimentary canal, it was expected that the percentage of assimilation of cesium for bluegill's food would be low. Cesium-137 on clay particles is probably bound so strongly that it is not released by the digestive juices in the stomach and gut.

The following experiment was designed to simulate the situation in White Oak Lake: A sample (100 g) of the uppermost bottom sediment from White Oak Lake was labeled with ^{137}Cs by mixing the sample with 150 ml White Oak Lake water on a magnetic mixer for 2 days after adding 125 μCi ^{137}Cs solution. After the mixing 94% of the tracer was bound by the sediments. Water was poured off and the "tagged" sediment sample was placed in a plastic jar and 3 ℓ of spring water were added. The concentration of ^{137}Cs in water per unit weight was 0.001 of that in the sediments, or the same as in White Oak Lake. Chironomus larvae were raised in a jar containing "tagged" sediments without any additional food. After the larvae had been living in sediments for 10 days, 30-45 larvae were fed to each of 16 bluegill (10 g). The initial quantities of ^{137}Cs in the fish were counted with a Packard Auto-Gamma 410 A single channel analyzer in conjunction with an Armac scintillation detector.

Fish were kept in a "Living Stream" unit with a continuous flow at 16 C. One or two fish were sampled after time intervals of 2 to 10 hours. After the fish were counted for ^{137}Cs, the stomach was cut open, and the contents were

TABLE 8

CONCENTRATION OF ^{137}CS IN DIFFERENT FOOD ITEMS OF WHITE OAK LAKE BLUEGILL

Date	Chironomid Larvae	Other Aquatic Insect Larvae	Algae	Terrestrial Insects	Roe	Detritus	Other Bottom Animals
Dec. 15, 1967	105.7					381.7	
Dec. 22, 1967	123.2	20.2	49.8				
March 28, 1968	55.2	18.4	23.4			190.4	
June 3, 1968			13.7		20.5	221.3	
July 16, 1968	72.4						39.7
Aug. 14, 1968				17.9			
Sept. 9, 1968		58.1	39.8				
Sept. 27, 1968	125.3		48.1			245.8	
$\bar{x}$	96.4	32.2	35.0	17.9	20.5	259.8	39.7
SD	28.0	18.3	14.2			73.0	

All values are given in pCi per gram fresh weight.

removed by scraping off all remains of food. The gut was emptied by stripping it several times through a pair of closed flap forceps and observing that no recognizable particles were left inside the gut. Stomach content, gut content, stomach, gut, and body were counted for ^{137}Cs with a dual-channel analyzer (Packard Tri-Carb 314 E) in conjunction with a well crystal (3" x 3") and an automatic sample changer (Packard Auto-Gamma).

The distribution of ^{137}Cs between the body, stomach and gut content is schematically shown in Fig. 5. The assimilation of ^{137}Cs occurred mostly in the stomach, and it was completed after 48 hours (Fig. 6). The mean value of assimilated ^{137}Cs was 7.1%. About 10% of the total quantity of ^{137}Cs assimilated was associated with the alimentary canal. Since all food remains were removed from the stomach and the gut, a relatively large proportion of assimilated ^{137}Cs seemed to be trapped by the walls of the alimentary canal.

This feeding experiment was conducted as a preliminary study using small bluegill, because they were more easily available in great numbers than large bluegill. Since the average weight of bluegill collected from White Oak Lake for ^{137}Cs, stable cesium, and potassium analyses was much higher, 107.6 g, the assimilation experiment was repeated using three groups of fish weighing 8 - 10 g, 18 - 20 g and 80 - 100 g, to see whether any differences occurred in the assimilation among different sizes. This time Chironomus larvae were kept for 4 weeks in tagged sediments to give them enough time to reach the equilibrium concentration of ^{137}Cs. The assimilation percentage in small bluegill was the same as in the previous experiment, but the size of fish affected the percentage

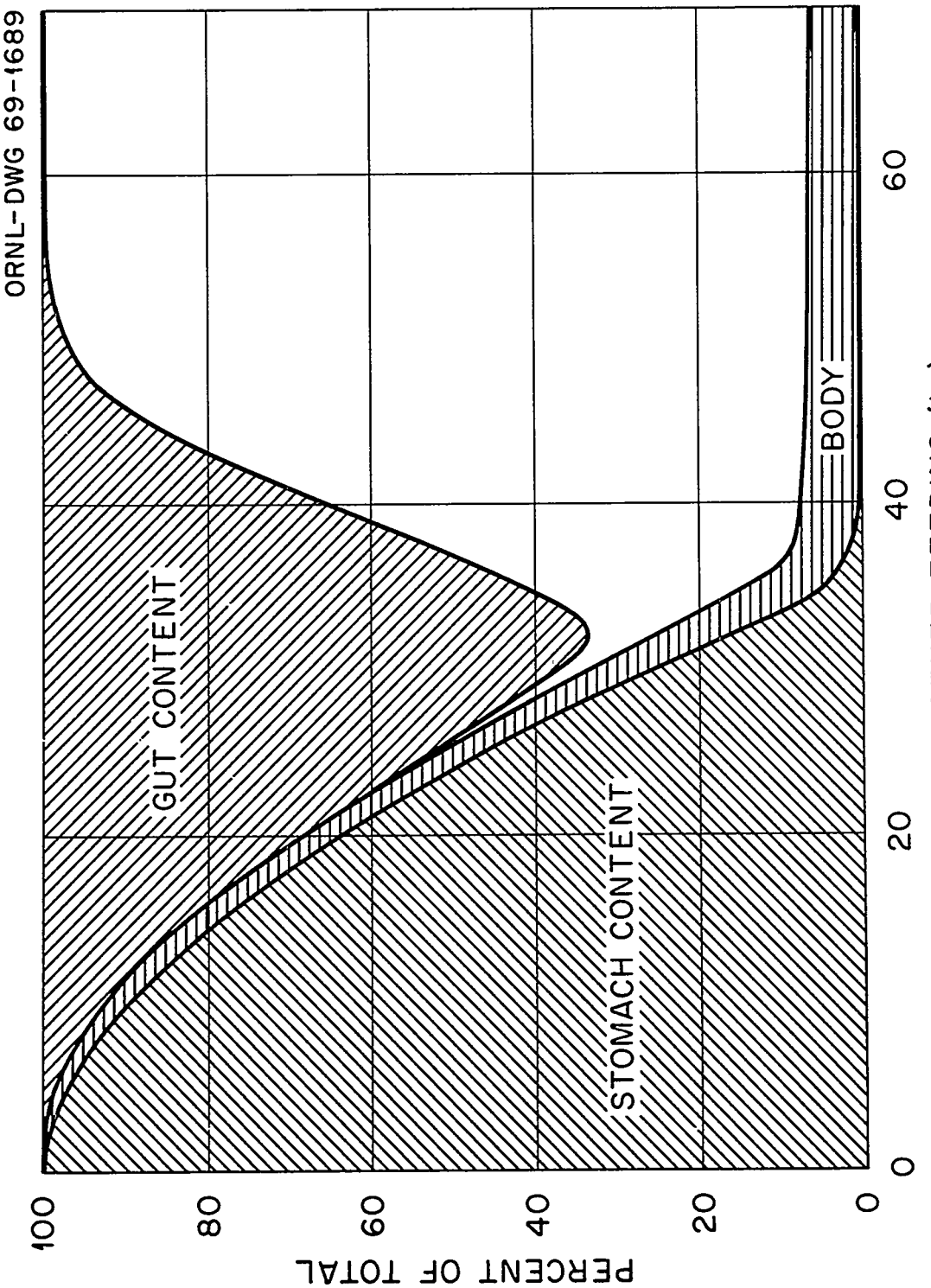

Figure 5. Distribution of ^{137}Cs in bluegill (10 g) after feeding a single meal of Chironomus larvae labeled in conditions similar to those in White Oak Lake.

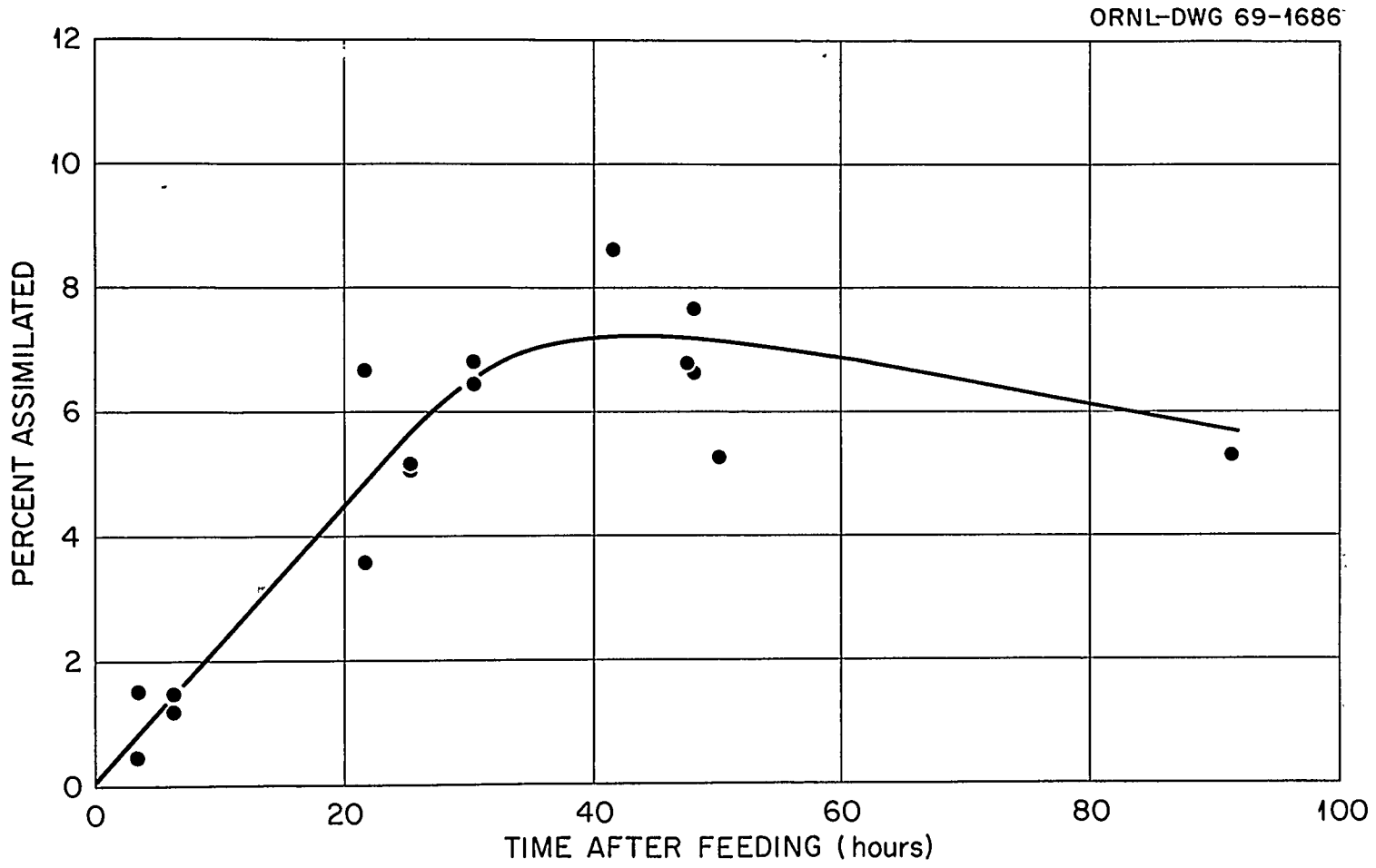

Figure 6. Assimilation percentage of ^{137}Cs in bluegill (10 g) after feeding a single meal of Chironomus larvae labeled in conditions similar to those in White Oak Lake.

of cesium assimilated from Chironomus larvae that lived in the conditions similar to those in White Oak Lake (Table 9). The differences in the assimilation among the different size groups were all significant at the level $P < 0.05$. The assimilation of ^{137}Cs for tagged detritus and algae was determined by feeding these food items in gelatin capsules to bluegill (80 - 100 g), and using the same counting procedure as above. The assimilation percentage for Chironomus larvae that had fed on algae containing ^{137}Cs was much higher than for Chironomus larvae that had fed on "tagged" detritus. This shows that bluegill can assimilate most of the ^{137}Cs associated with tissues, but they are able to assimilate very little of ^{137}Cs that is in the detritus and clay in the alimentary canal of Chironomus larvae. The assimilation percentage for detritus was only 3% (Table 9).

Daily Intake of ^{137}Cs

The body burden of ^{137}Cs in bluegill in White Oak Lake was in a non equilibrium state because the concentration of ^{137}Cs in large bluegill fluctuated 24% seasonally around the geometrical mean value (Fig. 2, p. 34); and in addition fish gained weight (20%) during the summer. If the calculated mean value of the radiocesium concentration in large bluegill (38.2 pCi/g fresh weight) is assumed to be the equilibrium value, the concentration of radiocesium in bluegill is evidently in a steady state with a noticeable seasonal cycling. Lotka (1956) defines the steady state not to be a true equilibrium in the dynamic sense, but a state maintained constant or approximately so with a continual

TABLE 9

PERCENTAGE OF ASSIMILATION OF ^{137}Cs FOR DIFFERENT TYPES OF FOOD ITEMS IN DIFFERENT SIZES OF BLUEGILL

Food Item	Weight of Bluegill							
	0.5-1.2 g		8-10 g		18-20 g		80-100 g	
	%	SD	%	SD	%	SD	%	SD
Food Items Similar to Those in White Oak Lake:								
Chironomus Larvae Fed on White Oak Lake Sediments			7.10	2.08	13.00	2.21	15.98	2.46
Algae							68.72	4.20
Detritus							3.01	0.21
Other Types of Food Items:								
Chironomus Larvae Fed on ^{137}Cs Containing Algae	34.0		68.6	2.2				
^{137}Cs Solution in a Gelatin Capsule							91.3	3.6

expenditure and dissipation or degradation of available energy. The body burden of radiocesium in an individual fish, however, will never reach a real steady state because a fish grows each year until its death, consequently, the body burden increases during its life span. Because of the seasonal cycling of the ^{137}Cs concentration and the continuous growth process of fish, equation (4.5, p. 30), was chosen for the calculations of the daily intake of ^{137}Cs. Since the biological half-life of ^{137}Cs in bluegill consists of two components, the excretion rate coefficient (k) in the equations (4.5, p. 30) had to be modified for this situation. For the loss term ($A_o e^{-kt}$) a weighted excretion rate coefficient (k') can be applied because (A_o) represents the body burden in the steady state when the proportions of the body burden excreted by T_{b_1}, (a_1), and by T_{b_2}, (a_2), stay practically constant. The weighted excretion rate coefficient can be calculated either on the basis of the proportions of the intake (p_1 and p_2) that are excreted by each rate or on the basis of the proportions of the body burden (a_1) and (a_2), that are excreted at each rate (see page 41). The weighted excretion coefficient (k') was calculated on the basis of the proportions of the intake (Kevern, 1966; Reichle, 1967):

$$k' = \frac{0.693}{T_{b_1} \cdot p_1 + T_{b_2} \cdot p_2} \quad , \qquad (4.16)$$

where T_{b_1} is the short component and T_{b_2} the long component of the biological half-life, and (p_1) and (p_2) the proportions of each assimilated intake excreted by those rates. The weighted excretion rate coefficient for bluegill above 70 g at the mean annual temperature of White Oak Lake (15.8 C) using the values

T_{b_1}, T_{b_2}, p_1 and p_2 in Table 6 (p. 44) was:

$$k' = \frac{0.693}{(7.6 \cdot 0.369) + (187.1 \cdot 0.631)} = 0.005734. \quad (4.17)$$

The intake can be subdivided into two intakes:

$$I = Ip_1 + Ip_2 \quad . \quad (4.18)$$

When these are substituted into the equation (4.5, p. 30) we get:

$$A_t = A_o e^{-k't} + \frac{Ip_1}{k_1}(1 - e^{-k_1 t}) + \frac{Ip_2}{k_2}(1 - e^{-k_2 t}) \quad . \quad (4.19)$$

In the intake part of the equation $\frac{I}{k}(1 - e^{-kt})$ it is assumed that (I), (k_1) and (k_2) stay constant during the time interval from t_o to t_t. Since the body burden of the White Oak Lake bluegill was changing considerably, a one-day time interval was chosen for calculations. So the value of (t) = 1 and (t) can be dropped from the equation. Now we can derive the equation (4.19) for the daily intake of ^{137}Cs:

$$I = \frac{A_t - A_{t-1} \cdot e^{-k'}}{\frac{p_1}{k_1}(1 - e^{-k_1}) + \frac{p_2}{k_2}(1 - e^{-k_2})} \quad . \quad (4.20)$$

The calculations were based on an average size fish that belonged to the age group III during the first part of the year and to the age group IV during the last part of the year. Since practically no growth occurred during the period from January to April, the weight of the fish in January was considered 91.6 g, the same as the weight of the fish when 4 years old. By next December this

fish had gained 18.9 g and weighed 110.5 g. The gain in weight was assumed to follow a sigmoid curve (Gerking, 1966) with the maximum growth occurring in May, June and July. Female fish lost about 7% of their weight while spawning and male fish about 1%. The concentration of ^{137}Cs in the gonads was 20.54 ± 0.68 (SD) pCi/g so the loss of the body burden of ^{137}Cs was only 4.5% in females and less than 1% in males. This loss was not taken into consideration in the calculations. The change in the weight of the bluegill (M) was simulated with an empirical fit to the following equation:

$$M = \arctan\,[0.02\ (t - 150)] \cdot 7.19 + 100.7\ g\ , \qquad (4.21)$$

where (t) is the number of the day of the year when half of the annual growth was achieved. The change in the concentration of ^{137}Cs/g fresh weight was simulated with a sine curve (Fig. 2, p. 34) and the concentration when multiplied by the weight (4.21) gave the body burden of ^{137}Cs in large bluegill at time (t):

$$A_t = \left\{\sin\left[\frac{2\pi}{360}\ (t-313)\right] \cdot 9.2 + 38.2\right\} \cdot \left\{\arctan\left[0.002 \cdot (t-150)\right] \cdot 7.19 + 100.7\right\} \qquad (4.22)$$

The body burden at time (t_o) was the value of (A_t) on the previous day, and so the formula for (A_{t-1}) was similar to that of (A_t) except that the time factors were (t-314) and (t-149).

The temperature fluctuation in White Oak Lake was simulated with a sine curve:

$$\phi = \sin\left[\frac{2\pi}{360}\ (t-105)\right] \cdot 10.8 + 15.8\ C \qquad (4.23)$$

with a maximum temperature 26.6 C in July and a minimum 5.0 C in January. The seasonal fluctuation of (k') was calculated by multiplying the value of (k')

at 15.8 C with the formula for $Q_{10} = 2$:

$$k'_t = 0.00573 \cdot e^{0.0693 \cdot \left\{\sin\left[\frac{2\pi}{360} \cdot (t-105)\right] \cdot 10.8\right\}} \qquad (4.24)$$

The excretion rate coefficients (k_1) and (k_2) were calculated using the values of T_{b1} and T_{b2} from Table 6:

$$k_1 = \frac{0.693}{7.6} = 0.0912 \qquad (4.25)$$

$$k_2 = \frac{0.693}{187.1} = 0.00370 \qquad (4.26)$$

The seasonal fluctuation of (k_1) and (k_2) was simulated using the same exponential function as for k' in the equation (4.24). Equations for growth (4.21), body burden of ^{137}Cs (4.22), excretion ratios (4.24 and two similar equations for k_1 and k_2), and the intake (4.20, p. 55) were calculated step-wise by a computer program.

The curves of growth, concentration of ^{137}Cs, body burden of ^{137}Cs, weighted excretion rate coefficient of ^{137}Cs and daily intake of ^{137}Cs are given in Fig. 7. The intake per gram of fish fluctuated from 0.065 pCi in late March to 0.334 pCi in the middle of August. The annual mean was 0.256 pCi.

Stable Cesium Balance in Bluegill

The concentration of stable cesium decreased somewhat with the increasing size of bluegill. In this respect the concentration of stable cesium differed from the concentration of radiocesium in bluegill (Tables 5, p. 33, and 10, p. 59). The concentration of stable cesium in bluegill over 70 g followed a seasonal pattern

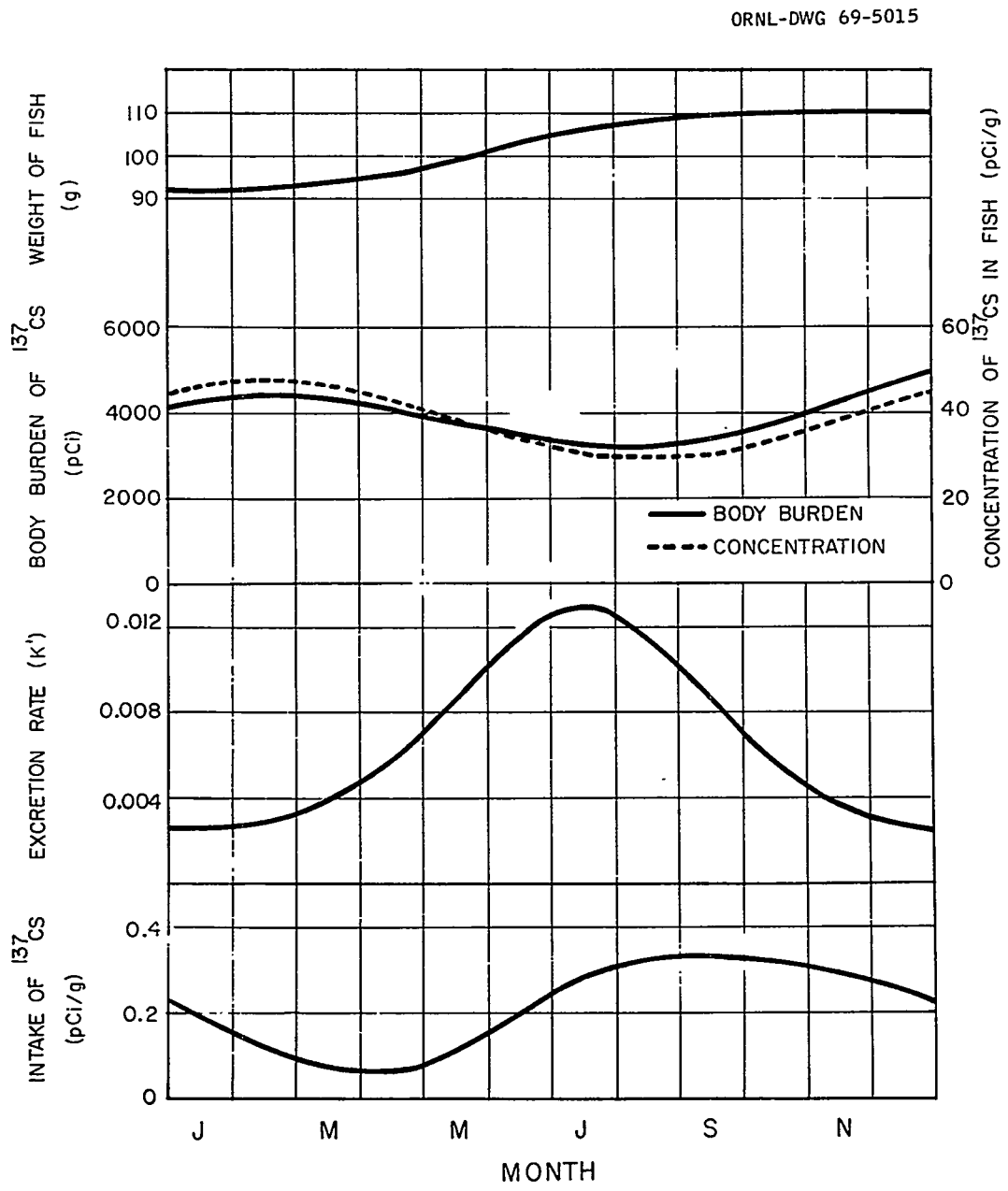

Figure 7. Calculated values of the weight, the concentration of ^{137}Cs, the body burden of ^{137}Cs, the weighted excretion rates (k'), and the daily intake of ^{137}Cs during a year for bluegill belonging to age-group III in January.

TABLE 10

CONCENTRATION OF STABLE CESIUM AND POTASSIUM IN BLUEGILL OF DIFFERENT SIZES

Weight g	Stable Cesium µg/g fresh wt Mean	SD	Potassium mg/g fresh wt Mean	SD	No. of Fish
0.5 - 1.5	.0101		3.37		19
10 - 20	.0096	.0035	3.07	0.19	9
20 - 40	.0089	.0030	3.05	0.11	16
40 - 70	.0094	.0024	2.95	0.13	4
> 70	.0089	.0023	2.73	0.24	154

similar to that of ^{137}Cs. The maximum, 0.0120 μg/g fresh weight, was in February and the minimum, 0.0058 μg/g fresh weight, in July (Fig. 8) with an annual mean value of 0.0089 μg/g fresh weight. The concentration of stable cesium in January 1968 was the same as reported by Nelson (1969) for the White Oak Lake bluegill in January 1966.

The biological half-life and the assimilation percentage of stable cesium was assumed to be the same as those for ^{137}Cs because organisms should not be able to differentiate between these two isotopes metabolically.

The quantities of stomach contents and Chironomus larvae were so small that it was necessary to combine all stomach contents together and all Chironomus larvae together to get enough material for stable cesium analysis. Therefore, the possible seasonal cycling of stable cesium concentration could not be determined in bluegill's food. The concentration of stable cesium in stomach contents was 0.0172 μg/g fresh weight. In Chironomus larvae the concentration was 0.0184 μg/g fresh weight.

The intake of stable cesium was calculated similarly to that of radiocesium using the equation (4.20). The intake varied from $2.80 \cdot 10^{-5}$ in January to $7.14 \cdot 10^{-5}$ μg/g of fish in August. The annual mean value was $4.79 \cdot 10^{-5}$ μg/g of fish.

Potassium Balance in Bluegill

The concentration of potassium decreased slightly with the increase in size of fish (Table 10) with concentrations of 3.37 mg/g in bluegill fingerlings

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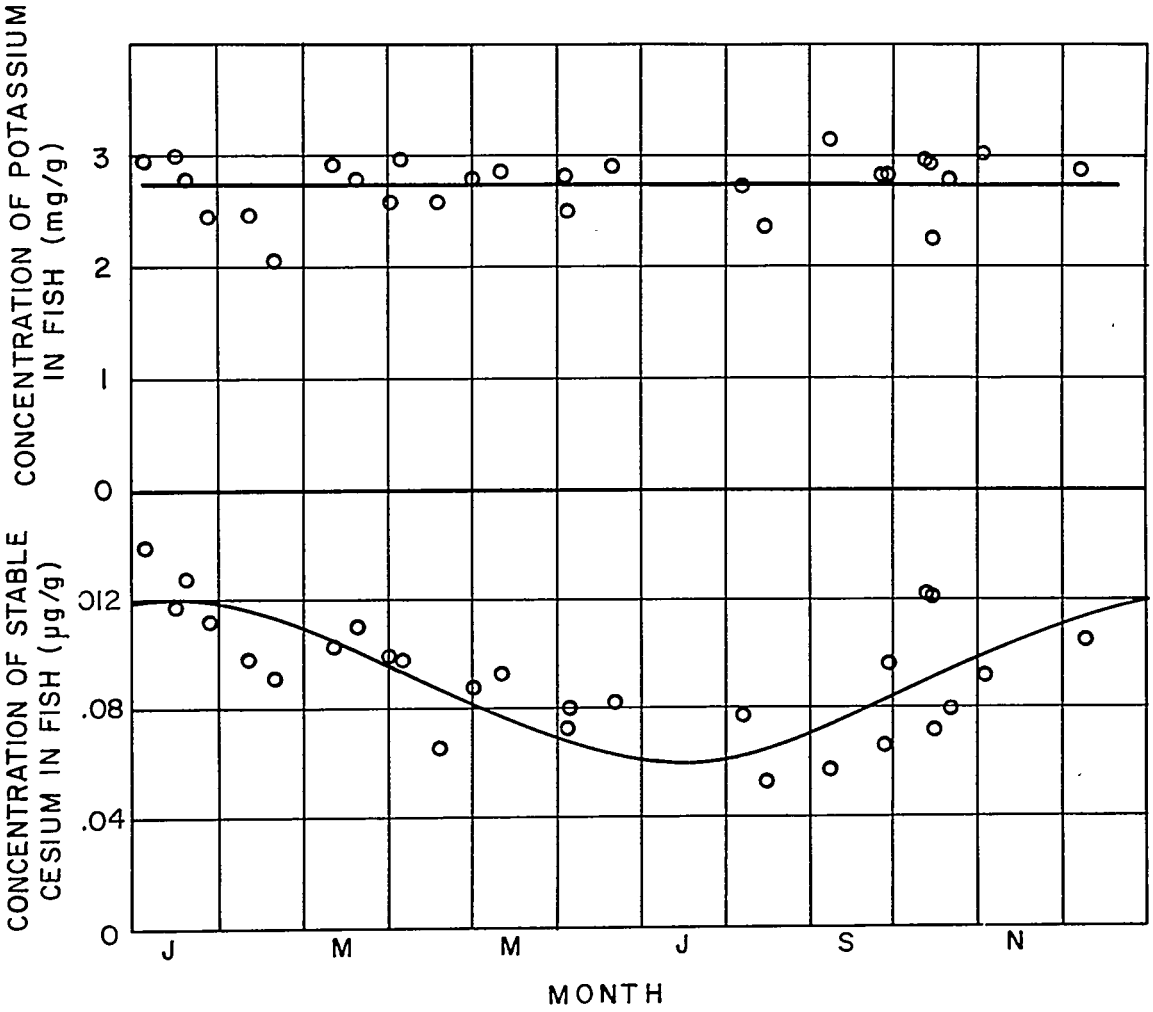

Figure 8. Concentration of potassium and stable cesium in bluegill over 70 grams.

compared to 2.73 mg/g in bluegill above 70 grams. Potassium concentrations in large bluegill did not fluctuate seasonally, but were constant throughout the year (Fig. 8).

Biological half-life experiments with ^{42}K required special designing because the physical half-life of ^{42}K is only 12.4 hours. Usually the T_b of a radioisotope in fish is determined by whole body counting. Whole body counting includes at least two types of errors, viz, a statistical error of the counting device and a geometrical error caused by small differences in the location of the sample in the counting chamber. These errors do not usually cause any problems while using long-lived radioisotopes. In experiments with ^{42}K the statistical error of the counting device increased rapidly while the radioisotope decayed and counting rates decreased. The counting error was about 10% on the fifth day of the experiments. The statistical error was about equal to the decrease in the body burden of ^{42}K in fish. This problem made it impossible to determine the T_b of ^{42}K by whole body counting. Therefore, the T_b of potassium was determined by analyzing the quantity of ^{42}K that was excreted by fish into water each day.

Errors associated with water counts were always less than 1% of the body burden. The geometrical error was also minimized because water could be counted more easily in a constant geometry than live fish and the counting times could be extended when the radioactivity had decreased. Live fish cannot be counted for a long time without a continuous water flow in the counting chamber. If the fish is placed into a container with a continuous water flow, either the geometrical

error is increased or the counting efficiency is lowered. The only disadvantage in the method used here was that fish had to be kept in 5 ℓ of water.

To avoid the disturbing effect of small space on the behavior and the physiology of fish, each fish was kept separately in a transparent plastic container which floated in a large aquarium. Five fish (50 - 70 g) were used in the experiment. The temperature was 17.0 ± 0.3 (SD) C.

The labeling of fish was performed by force-feeding ^{42}K in gelatin capsules. The volume of isotope solution was measured with a microsyringe (L. S. Starrett Co. No. 263 MRL). The fish were whole-body counted one hour after the feeding to check that fish had not regurgitated the capsule. Water in the containers was changed and counted every day. The counting was performed with a Packard 400-channel pulse height analyzer using a 3" x 3" NaI (Th) crystal. A Marinelli beaker with a volume of 5 ℓ was used for counting the water. The quantity of ^{42}K in the first day's water sample was subtracted from the initial body burden of the corresponding fish. On the next day the quantity of ^{42}K in water was again subtracted, but now from the value of the previous day, and so on.

The excretion was exponential, but the proportion of the short component was either very small or nonexistent (Fig. 9). The T_b of potassium was 37.0 ± 6.5 d (SD). No relationship was seen between the length of T_b of potassium and the weight of fish.

The uptake of potassium was studied by analyzing the stomach contents and food item samples for potassium and determining the assimilation percentage of potassium for different food items.

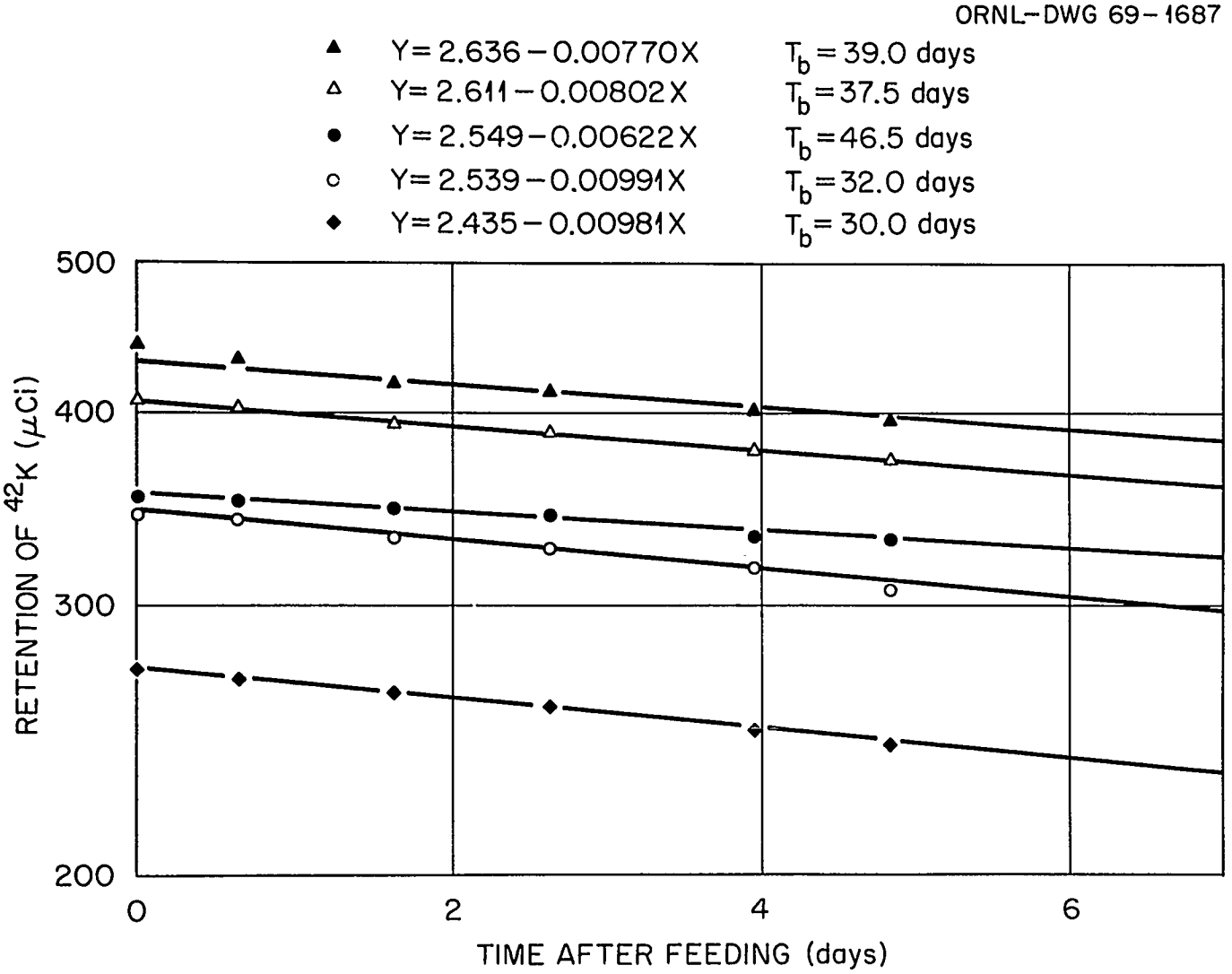

Figure 9. Retention and T_b of potassium after a single feeding of ^{42}K in five bluegill (50 - 70 g).

Potassium concentrations in the stomach contents and in Chironomus larvae were analyzed in the same combined samples as stable cesium. The concentration of potassium was 3.407 mg/g fresh weight in stomach samples, 2.846 mg/g in Chironomus larvae, and 8.80 mg/g in detritus.

The assimilation of potassium in bluegill for Chironomus larvae was determined with the same method used for the assimilation of ^{137}Cs. White Oak Lake sediments (100 g) were labeled with ^{42}K. Chironomus larvae were raised in the sediments for two days and then fed to 16 small bluegill (8 - 10 g)[1]. The assimilation percentage was 97.8 ± 1.2 (SD)% after 48 hours. Since the assimilation of potassium was almost complete in small bluegill, no experiments were conducted with larger bluegill. This percentage was applied to larger bluegill also. Potassium seemed to be deposited rapidly into muscle tissues because 80% of the tracer was found in the muscles 2 days after the inoculation. The distribution of tracer potassium in different tissues is given in Table II.

The intake of potassium was calculated in the same way as the intake of ^{137}Cs, but since the T_b of potassium had only one component, equation (4.6, p. 30) was applied. Potassium concentrations were constant throughout the year and therefore the body burden (A_t) was 2.73 mg times the weight at time (t). The excretion rate coefficient (k) was 0.0173 for the mean annual temperature (15.8 C). Potassium intake fluctuated from 0.0204 mg/g of fish in January to 0.105 mg/g of fish in July with an annual mean value of 0.0554 mg/g of fish.

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[1] The ability of sediments to sorb potassium was not as great as the ability to sorb cesium because only 60% of ^{42}K was in the sediments after two days' mixing.

TABLE II

DISTRIBUTION OF ^{42}K IN DIFFERENT ORGANS OF FOURTEEN BLUEGILL (8-11 g) 48 HOURS AFTER FEEDING ON CHIRONOMUS LARVAE THAT HAD BEEN RAISED IN WHITE OAK LAKE SEDIMENTS CONTAINING ^{42}K

Organ	Percent
Blood	2.70 ± 1.18[a]
Liver	3.69 ± 1.39
Kidney	2.15 ± 0.88
Stomach	3.54 ± 0.81
Gut	4.70 ± 1.08
Rest of the Body	81.00 ± 2.96
Stomach Content	0.60 ± 0.36
Gut Content	1.62 ± 1.06
	100.00

[a]All of blood could not be removed from the body, thus this value is an underestimate.

Calculation of Food Consumption in Bluegill

The quantity of assimilable ^{137}Cs in each stomach sample was calculated on the basis of the proportions of different food items present in the sample. Assimilation percentages for Chironomus larvae, vegetation, and detritus were taken from Table 9 (p. 53) while those for other aquatic insect larvae were assumed to be the same as for Chironomus since they also had detritus and clay in the gut and on the skin. The assimilation percentage for fish and roe was assumed to be 68.7%, the same as the assimilation for Chironomus larvae and algae not containing any detritus. Concentrations of ^{137}Cs in different food items were interpolated for different dates on the basis of the values in Table 8 (p. 48).

The concentration of ^{137}Cs analyzed in the stomach contents (Table 7, p. 46) agreed well with the values that were calculated (Table 12) using the proportions of each food item in the stomach contents (f) and the interpolated concentrations of ^{137}Cs in each food item (d). Values of I were taken from Fig. 7 (p. 46) and intake of food was calculated with the equation:

$$r' = \frac{I}{\Sigma a_i d_i f_i} \qquad (4.27)$$

For example, on the 4th of June 1968, the calculated ^{137}Cs intake, I, was 0.168 pCi/g fish. The stomach contents of seven bluegills consisted of: Chironomid larvae, 50%; other insect larvae, 20%; plants, 11%; and detritus, 19%. The assimilated quantity of ^{137}Cs for food was calculated in the following way:

TABLE 12

CALCULATED VALUES OF ^{137}CS CONCENTRATION IN STOMACH CONTENTS ($\Sigma d_i f_i$), DAILY ASSIMILATED INTAKE OF ^{137}CS (I), AND THE QUANTITY OF ^{137}CS ASSIMILATED FROM ONE GRAM OF STOMACH CONTENTS ($\Sigma a_i d_i f_i$)

Date	^{137}Cs in Stomach Contents $\Sigma d_i f_i$ pCi/g	I pCi/g fish	$\Sigma a_i d_i f_i$ pCi/g	No. of Stomach Samples
Sept. 29, 1967	74.0	0.333	18.0	11
Nov. 2, 1967	102.5	0.332	19.0	3
Dec. 1, 1967	101.0	0.283	21.9	4
Jan. 4, 1968	160.5	0.216	20.7	4
Feb. 20, 1968	115.5	0.111	13.3	3
March 11, 1968	70.1	0.076	7.5	2
April 5, 1968	84.0	0.065	6.2	14
April 18, 1968	65.7	0.071	5.0	5
June 4, 1968	69.0	0.168	6.4	7
June 20, 1968	58.5	0.220	6.8	4
July 26, 1968	59.1	0.305	11.5	2
Aug. 6, 1968	60.7	0.315	14.0	12
Aug. 14, 1968	17.9	0.323	12.2	4
Sept. 7, 1968	81.1	0.333	19.7	5
Sept. 27, 1968	79.5	0.333	27.4	5
$\bar{x}$	80.7	0.233	13.5	Σ 84
SD	30.0	0.106	6.7	

All values are given for fresh weight.

Food Item	a_i %	d_i pCi/g	f_i %	$a_i d_i f_i$ pCi
Chironomid Larvae	16	40	50	3.2
Other Insect Larvae	16	28	20	0.9
Plants	69	14	11	1.0
Detritus	3	221	19	1.3
				Σ 6.4

The sum of $d_i \cdot f_i$ was 69 pCi or the same as analyzed by γ-counting (68 pCi) (Table 7, p. 46).

The maximum values of assimilated ^{137}Cs occurred in fall and the minimum values in spring (Fig. 7, p. 58). This seasonal fluctuation of the assimilated quantity was primarily caused by the fluctuation of the concentration of ^{137}Cs in the food items, although the seasonal changes in the quality of food also partly affected the quantity of assimilated ^{137}Cs.

The daily meal, or the quantity of food in percent of the body weight, fluctuated from 0.84% in February to 3.24% in June with a mean value for the whole year of 1.75% (Fig. 10). The correlation of the feeding rate in bluegill with water temperature in White Oak Lake followed the linear regression $Y = 0.32 + 0.081\ X$ ($r^2 = 0.665$) where Y is the daily meal in percent of body weight and X the temperature in C.

Food consumption was also calculated with the balances of stable cesium and potassium. Stable cesium intake was calculated with equation (4.20, p. 55), and the weighted assimilation percentage of cesium, 15.2%, was calculated

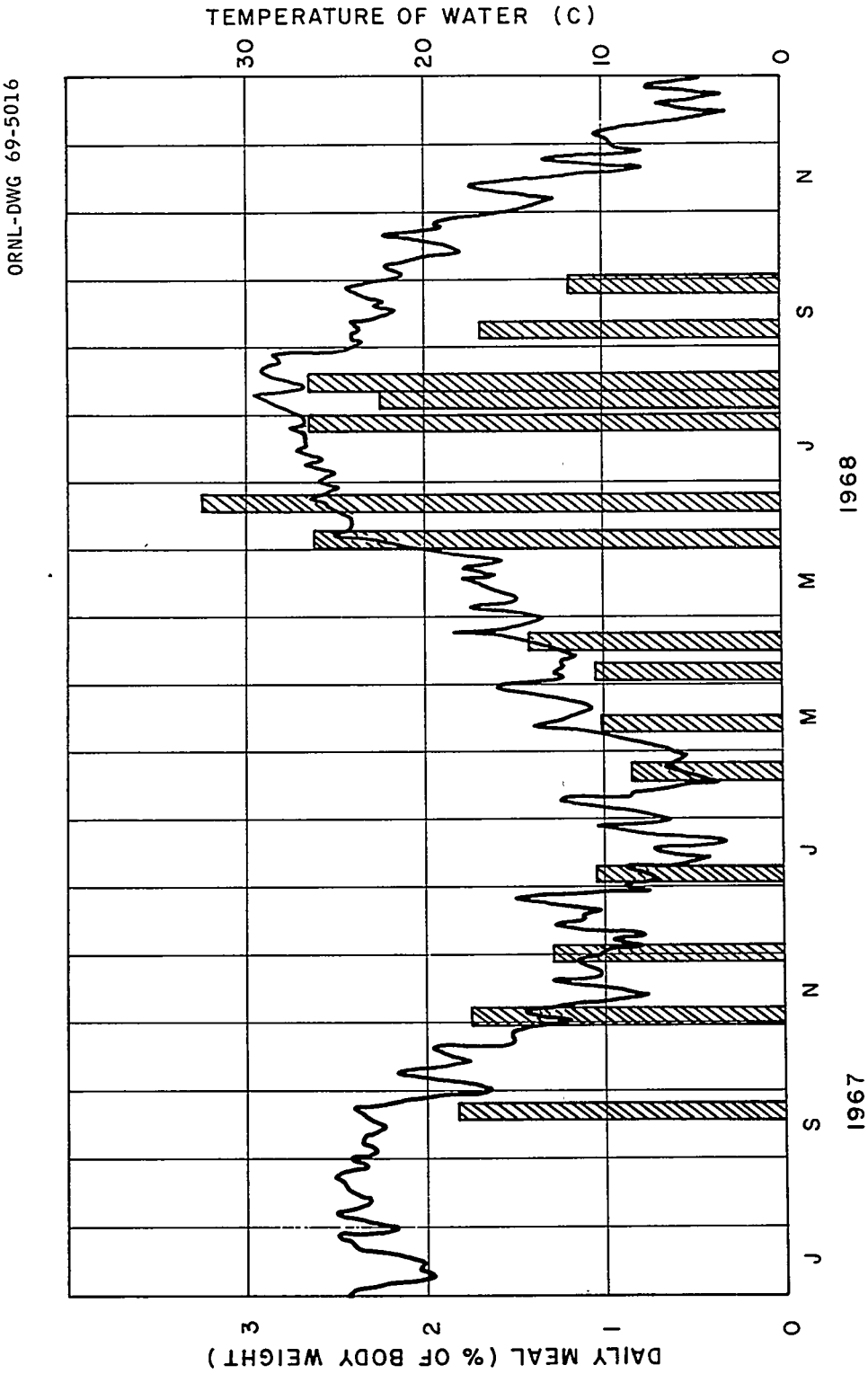

Figure 10. Daily meal of bluegill belonging to the age-group III in January and the temperature of water in White Oak Lake.

for the whole year on the basis of the stomach analyses with the relationship:

$$a' = \frac{\Sigma a_i d_i f_i}{\Sigma d_i f_i} \cdot 100 \quad . \qquad (4.28)$$

The concentration of stable cesium in the combined stomach samples was 0.0151 µg/g fresh weight and the intake of food was accordingly:

$$r' = \frac{0.0000479}{0.0151 \cdot 0.152} = 0.0209 \text{ g/g of fish} \quad . \qquad (4.29)$$

The mean annual daily meal was 2.09% of body weight on the basis of stable cesium balance. This value was somewhat higher than the one calculated using the balance of ^{137}Cs. The difference might reflect differences in the assimilation between ^{137}Cs and stable cesium. Also the fact that stable cesium was analyzed only in one combined sample that consisted of all stomach samples may have caused part of the difference.

Potassium assimilation was 97.8% for Chironomus larvae and it was considered the same for all organisms. For detritus the assimilation was 72%. The concentration of potassium in the stomach contents was 3.407 mg/g fresh weight, in Chironomus larvae 2.846 mg/g and in detritus 8.8 mg/g. Since stomach contents consisted of 9.6% detritus, the potassium concentration in organisms was $3.407 - (0.096 \cdot 8.8) = 2.562$ mg K/g. The food intake of bluegill on the basis of potassium balance was:

$$r' = \frac{0.05546}{0.978 \cdot 2.562 \cdot 0.904 + 0.72 \cdot 8.8 \cdot 0.096} = 0.0193 \text{ g/g of fish} \quad . \qquad (4.30)$$

The daily meal, 1.93% of body weight is again slightly higher than that determined with the uptake of ^{137}Cs. One possible source of error was that the biological

half-life of potassium may have been an underestimate. It may also be that the T_b of potassium does not follow the relationship $Q_{10} = 2$.

Concentrations of ^{137}Cs, Stable Cesium and Potassium in Different Tissues of Bluegill

Tissues of 12 bluegill were combined as one sample and analyzed for ^{137}Cs, stable cesium and potassium in the same way as fish samples. Flesh had a high concentration of ^{137}Cs and potassium, but skin had the highest concentration of stable cesium. Ovaries had low concentrations of ^{137}Cs, stable cesium and potassium (Table 13). The entrails, excluding liver and genitals, had the highest concentration of ^{137}Cs, but the lowest concentration of stable cesium and potassium. Reasons why the rest of the entrails (mainly alimentary canal) had much more ^{137}Cs than stable cesium were not known. Concentrations of stable cesium and potassium in different tissues did not follow the same order as in the Clinch River bluegill since flesh of Clinch River bluegills had a lower concentration of these elements than liver and ovaries (Nelson, 1969). The specific activity values of bluegill tissues in White Oak Lake varied somewhat between different tissues, but the value of the entrails was much higher than that of individual tissues. The specific activity among the individual tissues studied was highest in flesh and lowest in ovaries.

TABLE 13

CONCENTRATIONS OF ^{137}CS, STABLE CESIUM, AND POTASSIUM AND SPECIFIC ACTIVITIES OF ^{137}CS IN DIFFERENT TISSUES OF BLUEGILL

Tissue	^{137}Cs pCi/g fresh wt	Stable Cesium µg/g fresh wt	Potassium mg/g fresh wt	Specific Activity pCi/µg Cs
Flesh	42.70	.0078	3.36	5520
Liver	40.44	.0077	2.63	5240
Ovaries	20.54	.0056	2.20	3640
Rest of Entrails	53.55	.0033	1.33	10600
Skin	31.91	.0085	2.59	3770
Whole Body	35.60	.0072	2.68	4950

Values based on twelve fish collected June 3, 1968.

CHAPTER V

CONCENTRATIONS OF ^{137}CS, STABLE CESIUM AND POTASSIUM AND SPECIFIC ACTIVITIES OF ^{137}CS IN DIFFERENT SPECIES OF FISH IN WHITE OAK LAKE

The concentration of ^{137}Cs in different species of fish (Table 14) did not follow the trophic level relationship (see p. 16). Gizzard shad and golden shiner that are partly primary consumers, feeding on plankton and algae, had high concentrations of ^{137}Cs. Piscivorous largemouth bass had also a high ^{137}Cs concentration. Warmouth, even though piscivorous, had a low concentration. Lowest concentrations were in the omnivorous redear. Kevern and Griffith (1966) reported a higher concentration of ^{137}Cs in bluegill (38 pCi/g) than in gizzard shad (32 pCi/g) and in largemouth bass (35 pCi/g) in White Oak Lake in January 1966.

Part of these interspecific differences was caused by the fact that ^{137}Cs concentrations were based on mean values of all samples analyzed. Warmouth samples, e.g., were primarily from the time of the year when the concentration of ^{137}Cs in fishes was at a minimum because of the seasonal cycling. Golden shiner samples were mostly from the time when the concentration was at a maximum.

The body burden of ^{137}Cs is determined by the rates of intake and excretion of ^{137}Cs. There are four factors: (1) assimilation, (2) feeding rate, (3) concentration of ^{137}Cs in food, and (4) biological half-life, that could cause

TABLE 14

CONCENTRATION OF ^{137}CS IN WHITE OAK LAKE FISHES

Species	^{137}Cs pCi/g fresh wt	SD	No. of Fish
Gizzard Shad	47.03	12.16	15
Golden Shiner	62.61	22.05	15
Goldfish	34.53	13.86	10
Redear Sunfish	26.85	5.86	40
Bluegill	40.12	15.21	186
Warmouth	36.69	15.75	37
Largemouth Bass	52.75	13.14	6

the differences. Assimilation of ^{137}Cs in all fish species is probably between 60% and 80% for food not containing any clay. This means that the fishes, like gizzard shad and golden shiner, that feed partly on plankton and periphyton, and piscivorous fish, like warmouth and largemouth bass, had a high percentage of assimilation while omnivorous redear sunfish and bluegill, feeding mostly on bottom animals, had the lowest assimilation percentages. Goldfish probably had feeding habits similar to those of carp (Kevern, 1966) and ate algae and detritus, and consequently, also had high assimilation percentage.

Feeding rates of different species were not studied, but some remarks on the concentration of ^{137}Cs in different food items can be made. The concentration of ^{137}Cs in algae was 35.0 pCi/g fresh weight on the average, and considering 70% assimilation, 25 pCi of ^{137}Cs would have been assimilated from 1 g of algae. Assimilation of ^{137}Cs from Chironomus larvae was only 15.4 pCi/g (96.36 pCi · 0.16). The concentration of ^{137}Cs in fish fingerlings was 10.5 pCi/g which means that small, piscivorous fish assimilated only 7.4 pCi/g (10.5 pCi · 0.70) of fish consumed while the assimilation of ^{137}Cs/g unit weight of food in plankton and periphyton feeders was much higher. Biological half-lives of ^{137}Cs in different species were not studied, but since the biological half-life varied by a factor of four in different species at the same temperature (Hä̈sänen et al. 1967), the excretion rates may contribute partly to the differences in the concentration of ^{137}Cs among the species in White Oak Lake.

High concentrations of ^{137}Cs in golden shiner and gizzard shad can be explained by the assimilation percentage and the concentration of ^{137}Cs in

food. The low concentration of ^{137}Cs in fish fingerlings might explain the low concentration of ^{137}Cs in warmouth. Largemouth bass ate bigger fish than warmouth and since large fish had a higher concentration of ^{137}Cs than fingerlings, this in turn could explain the differences between the concentration of ^{137}Cs in these two piscivorous fish.

The concentration of ^{137}Cs in fishes of non-turbid and in slightly turbid lakes followed the "Trophic level effect" (Kolehmainen et al. 1964, 1966, 1967, 1968c), but in White Oak Lake clay particles changed the efficiency of ^{137}Cs assimilation so much that ^{137}Cs was not equally available for fish at different trophic levels.

Stable cesium concentrations in White Oak Lake fish did not show a clear relationship with the concentration of ^{137}Cs (Table 15). The concentration of stable cesium was highest in goldfish and lowest in bluegill and values of stable cesium in gizzard shad, bluegill and largemouth bass were on the same level as determined by Nelson (1969) in January 1966.

Specific activities of ^{137}Cs varied in different species by a factor of two, being lowest in goldfish (2000 pCi/μg cesium) and highest in golden shiner (5250 pCi/μg cesium) (Table 15). Somehow goldfish were able to concentrate stable cesium more than radiocesium while golden shiners were able to concentrate more ^{137}Cs than stable cesium. The specific activity of water was 2050 pCi/μg stable cesium and specific activities of fishes were 1.0 to 2.5 times that of water. Since these differences in the specific activities were present in all samples analyzed, it is apparent that these were real differences and were

TABLE 15

CONCENTRATIONS OF ^{137}CS, STABLE CESIUM, AND POTASSIUM AND SPECIFIC ACTIVITIES OF ^{137}CS IN WHITE OAK LAKE FISHES

Species	^{137}Cs pCi/g Mean	SD	Stable Cesium Mean	SD	Potassium Mean	SD	Specific Activity pCi/μg Cs	No. of Fish
Gizzard Shad	42.49	2.84	.0115	.0034	3.00	0.14	3710	5
Golden Shiner	76.06		.0145		2.77		5250	4
Goldfish	38.97	8.41	.0195	.0085	2.18	0.37	2000	9
Redear Sunfish	25.29	5.82	.0090	.0022	2.81	0.32	2830	29
Bluegill	40.20	15.23	.0089	.0023	2.73	0.24	4550	154
Warmouth	28.82	11.03	.0121	.0031	2.88	0.10	2380	24
Largemouth Bass	58.55	6.54	.0175	.0023	3.16	0.65	3340	2
Water[a]	58.2	25.6	.0284	.0130	1.77	0.07	2050	3

[a]Values are given per liter.

not caused by analytical errors. Organisms should not be able to distinguish between ^{137}Cs and stable cesium, thus the differences in the specific activities must be caused by differences in the availability of ^{137}Cs and stable cesium somewhere along the foodchains.

The concentration of stable cesium in Clinch River bluegill was 0.0052 µg/g ± 0.0028 (SD) (n = 8) on the 4th of May, 1968, which is lower than the concentration in the White Oak Lake bluegill. Nelson (1969) reported a lower value of stable cesium, 0.034 µg/g, for flesh of bluegill in the Clinch River.

Potassium concentrations were rather uniform in all species. The lowest concentration was in goldfish, 2.18 mg/g fresh weight, and the highest in largemouth bass, 3.16 mg/g fresh weight (Table 15). There was no relationship between the concentration of potassium and radiocesium or between potassium and stable cesium. The concentration of potassium in the Clinch River bluegill was 2.74 mg/g or the same as the average in the White Oak Lake bluegill.

The concentration factors of ^{137}Cs and stable cesium varied by a factor of two among different species, but C.F. of potassium was rather uniform (Table 16). The C.F.'s of ^{137}Cs in White Oak Lake were on the same level as C.F.'s of fallout ^{137}Cs in lakes that had the same concentration of potassium in water (Bortoli et al. 1967b; Kolehmainen et al. 1967; Preston et al. 1967). Concentration factors of stable cesium in White Oak Lake fish were similar to those determined by Nelson (1969) and they also were of the same level as in three species of fish in Lake Comabbio in northern Italy that had a potassium

TABLE 16

CONCENTRATION FACTORS OF ^{137}Cs, STABLE CESIUM AND POTASSIUM IN WHITE OAK LAKE FISHES

Species	^{137}Cs	Stable Cesium	Potassium
Gizzard Shad	810	400	1690
Golden Shiner	1080	510	1570
Goldfish	530	690	1230
Redear Sunfish	460	320	1590
Bluegill	710	310	1540
Warmouth	630	430	1640
Largemouth Bass	910	620	1770

concentration of 1.6 mg/ℓ (Bortoli et al. 1967). Concentration factors of potassium in White Oak Lake fishes were the same as in other lakes with the same concentration of potassium in water (Fig. 1, p. 7).

CHAPTER VI

DISCUSSION

Comparison of the Balances of ^{137}Cs, Stable Cesium and Potassium in the White Oak Lake Bluegill

The uptake rate of cesium in fish is affected by the concentration of the radioisotope in food, the percentage of assimilation, and the rate of food consumption. The concentration of ^{137}Cs in the food organisms of fish is directly proportional to the concentration of ^{137}Cs in water and inversely related to the concentration of potassium in water (pp. 13-14). This is probably also true for stable cesium. Potassium concentration in food organisms of fish are, however, independent of the concentration of potassium in water (p. 4) and are rather uniform in all organisms. In White Oak Lake, the food organisms of bluegill had a seasonal cycling of ^{137}Cs concentration with a maximum concentration in winter and a minimum in summer. The seasonal pattern of stable cesium and potassium concentration in bluegill's food items was not studied, but stable cesium concentrations might have followed a similar cycling as ^{137}Cs concentrations. Potassium concentrations in food apparently stayed rather uniform throughout the year.

The assimilation percentage of ^{137}Cs in carp varied for different food items in White Oak Lake (Kevern, 1966) and assimilation also varied for different

food items in the White Oak Lake bluegill. Assimilation was low for food that contained clay particles such as Chironomus larvae (16%) and detritus (3%), but the assimilation was about 70% for Chironomus larvae and algae which did not contain clay particles. The assimilation percentage increased with the weight of fish such that assimilation from Chironomus larvae not containing clay was 34% in bluegill fingerlings (0.5 - 1.2 g) while it was 68% in bluegill of 10 g. In 10-g bluegills the assimilation for White Oak Lake Chironomus larvae (with detritus in alimentary canal) was 7% while it was 16% in bluegill of 100 g. The reason for the increase of the assimilation efficiency with the size of fish is not known. However, the passage of food through the alimentary canal was completed in fingerlings in about 12 hours while it took about 60 hours in bluegills of 100 g at the same temperature. It may be that the assimilation is related to the time of digestion. There might also be differences in the composition of digestive juices among different sizes of bluegills.

The assimilation percentage of potassium (98% for White Oak Lake Chironomus larvae) was higher than that of cesium. Potassium assimilation seemed to be more complete for all types of food than for the assimilation of ^{137}Cs. Even for detritus the assimilation of potassium (72%) was high but assimilation of ^{137}Cs from detritus was only 3%.

The rate of food consumption decreases with increasing fish size (Minckley et al. 1963; Winberg, 1956). This means that the ingestion rate of ^{137}Cs, stable cesium and potassium decreased with increasing fish size in White Oak Lake. However, the intake of ^{137}Cs in bluegill fingerlings was about twice of that

in bluegill over 70 g per unit weight when calculated on the basis of the body burden and the excretion rates. Bluegill fingerlings had a diet similar to that in large bluegills, and since the assimilation of ^{137}Cs in bluegill fingerlings was only one fourth of that in large bluegills, the food intake in bluegill fingerlings was six to eight times the intake of food in large bluegills per unit weight.

The concentration of ^{137}Cs fluctuated both seasonally and yearly in a fallout situation (Kolehmainen et al. 1967, 1968a,b,c; Bortoli et al. 1967b; Gustafson, 1969). Nelson (1969) reported changes in the concentration of ^{137}Cs and stable cesium in Clinch River crappie. The seasonal cycling of ^{137}Cs and stable cesium in bluegill and other fishes in White Oak Lake was noticeable, with a maximum in the winter and a minimum in the summer (pp. 32 and 60). Pendleton (1959) reported low contamination levels of ^{137}Cs during both high and low temperature extremes in pumpkinseed (Lepomis gibbosus) in a concrete pond experiment, but on the basis of the figure in the same publication, it seems very likely that the pattern of the seasonal fluctuation of ^{137}Cs in pumpkinseed was almost the same as that in the White Oak Lake bluegill. The concentration of ^{137}Cs in the Par Pond bluegill (Harvey, 1964) did not show the same type of seasonal fluctuation as in White Oak Lake bluegill. Kevern (1966) studied the balance of ^{137}Cs in the White Oak Lake carp, but he did not have enough samples to show any seasonal trend in the concentration, and he assumed the concentration to be in equilibrium.

The concentration of ^{137}Cs in White Oak Lake bluegill increased with fish size up to 70 g when it reached the steady state. In the fallout situation the concentration of ^{137}Cs also increased with the fish size (Hannerz, 1966). The concentration of stable cesium in the White Oak Lake bluegill did not increase, but rather decreased slightly with increasing fish size.

Potassium concentrations also decreased slightly with increasing fish size, but there were no seasonal changes in concentrations. The concentration of potassium in bluegills was considered to be in equilibrium. Potassium concentration in all species of fresh water fish are paractically constant (Kolehmainen et al. 1966, 1968b; Nelson, 1969). In pacific salmon (Oncorhynchus tschawytscha) MacLeod et al. (1958) and in sockey salmon (Oncorhynchus nerka) Tomlinson et al. (1967) found a small drop in the concentration of potassium during sexual development and spawning.

The body burden of fish does not reach equilibrium for ^{137}Cs, stable cesium or even for potassium because fish grow every year throughout their lifespan and consequently the body burdens of all elements are increasing. Growth of fish is not continuous but the gain in weight occurs mostly in spring (Brown, 1957; Gerking, 1962; Gross et al. 1963) while the weight may, in some cases, lessen during the winter. Kevern (1966) assumed the body burden of ^{137}Cs in yearling carp in White Oak Lake consisted of two pools, viz, ^{137}Cs in old tissues, and ^{137}Cs in new tissues that were produced during the second-year's growth. He assumed the old tissues were in equilibrium with ^{137}Cs in food and calculated separately the intake of ^{137}Cs that was required to keep

the concentration of ^{137}Cs in new tissues the same as in old tissues. Kevern's method is generally not applicable because he treated the growth of carp as a continuous linear process which is not true in fish. The calculations of the balances of ^{137}Cs, stable cesium and potassium in bluegill were based on an equation that was derived for a non-equilibrium situation (see p. 30). The calculations took into account the seasonal cycling of the concentration, and variations in growth rates and the excretion rates. Values for the intake of ^{137}Cs, stable cesium and potassium were calculated on a daily basis throughout the year instead of using annual or monthly mean values as had been done previously (Kevern, 1966; Kolehmainen et al. 1967).

The long component of biological half-life of ^{137}Cs increased with the size of bluegill as in other fishes (Morgan, 1964; Häsänen et al. 1967, 1968). The effect of temperature was not studied, but it was assumed the T_b doubled for a drop of 10 C (Kevern et al. 1964a; Kevern, 1966; Häsänen et al. 1967, 1968). Temperature relationships of the biological half-life of ^{137}Cs have not yet been studied for the biokinetic range of any species of fish so that it is not known in which temperature range, if any, T_b actually follows the Q_{10} law. Even if this were determined in the laboratory, it is not sure whether it would be valid for nature. The Q_{10} relationship is based on the fact that the rate of chemical reactions increases two to three times when the temperature increases 10 C. Metabolism of fish tends to change during the gradual change in temperature by the adaptation processes, and consequently, the metabolism of fish does not follow exactly the Q_{10} law. For example, oxygen consumption of bluegill in nature

was higher in winter than was expected on the basis of the relationship with the temperature alone (Wohlschlag and Juliano, 1959).

The biological half-life of potassium in bluegill (60 g) was 40.1 d at 15.8 C. This is about the same value as in man (Pendleton et al. 1965). The ratio between T_b of potassium and ^{137}Cs was 1 : 4.5 in bluegill while it was 1 : 3 in man (Pendleton et al. 1965; Fujita et al. 1966). Retention curves of potassium in bluegill did not show any short component, but a short component may have been present if the retention of ^{42}K could have been followed longer. The percentage of the short component after a single meal must have been small because the fit of the data points with the one-component exponential curve was excellent (Fig. 9, p. 64). In mice and rats the proportion of T_{b_1} was 2% after a single feeding of ^{42}K (Richmond, 1958). If bluegill actually excreted a proportion of the assimilated quantity of ^{42}K by a short component, calculated T_b's assuming only one component would be underestimated (Fig. 9, p. 64).

Specific Activity and Equilibrium Concepts in White Oak Lake Fish

The specific activity concept has been used successfully to predict the concentrations of the radioisotope in aquatic organisms in the environments that receive a chronic input of radioactive wastes (Nelson, 1967, 1969; Kaye and Nelson, 1968). The use of specific activity implies that the organisms are in equilibrium with the environment. In White Oak Lake bluegills the concentration of both ^{137}Cs and stable cesium had a similar seasonal cycling, and so the specific activity varied only slightly with time. Since the ^{137}Cs concentration

fluctuated 24% and stable cesium 34% around the mean value (pp. 32 and 60), the concentrations of ^{137}Cs and stable cesium were not in equilibrium in bluegill. If the specific activity in fish is the same as in water, it does not necessarily mean that fish are in equilibrium with the radioisotope in the environment. Before the specific activity concept can be applied to prove that the organisms are in equilibrium with the radioisotope in the environment, one has to show that the organisms are in equilibrium with the concentration of the stable element in the environment.

The specific activity in different species of fish varied by a factor of two (p. 77). These differences are a contradiction of the statement that ^{137}Cs and stable cesium behave similarly in aquatic environments (Nelson, 1969). One possible reason for the differences may be that stable cesium and ^{137}Cs are not equally available for different species of fish. This would mean that stable cesium and ^{137}Cs had different physicochemical or biological behavior in White Oak Lake which in turn would make the specific activity concept less useful in predicting ^{137}Cs concentrations (Kaye and Nelson, 1968).

The specific activity of ^{137}Cs in the White Oak Lake bluegill was twice that of White Oak Lake water. Bortoli et al. (1967b) found that the concentration factors of radiocesium were four times higher than the concentration factors of stable cesium in three species of fish in northern Italian lakes which means that the specific activity in fish was four times as high as that of water. Stable cesium was analyzed by Bortoli et al. (1967b) with neutron activation using uncentrifuged water samples that included all plankton and tripton. Their values

were, therefore, total stable cesium concentrations which are always higher than those of dissolved cesium. The concentration of stable cesium in White Oak Lake water, however, was analyzed as dissolved stable cesium. Since the specific activities of ^{137}Cs in most of White Oak Lake fishes were higher than the specific activity in water, ^{137}Cs seemed to be more easily available for organisms than stable cesium in White Oak Lake. The values of stable cesium concentration in water were based only on three water samples and the standard deviation was great. Therefore, it is not known whether there was a seasonal cycling of the concentration of stable cesium in water or whether the differences were caused by random variation.

Radioisotope Method in Determining the Food Consumption Rate in Fish

The balance of a radioisotope is one practical method to determine the feeding rate of animals both in terrestrial and aquatic ecosystems. This method was first used to calculate the food consumption in fish that were in equilibrium with their environment (Davis and Foster, 1958; Foster, 1959). Afterwards the radioisotope method has been applied to study the feeding rates and foodweb relationships among terrestrial arthropods (Reichle and Crossley, 1965; Crossley, 1966; Reichle, 1967) and some work has been done in aquatic ecosystems (Kevern, 1966; Kolehmainen et al. 1967). Radiophosphorus was used by Podoliak (1961) to study the uptake of phosphorus by fish. Nelson and Malone (1968) used a simplified radioisotope method to study the feeding rate of aquatic snails.

In the feeding rate experiments, the rates of intake and excretion have

usually been treated as constants and an equilibrium situation has been assumed during the time of the experiment (Reichle and Crossley, 1965, 1969; Crossley, 1966; Reichle, 1967; Williams and Reichle, 1968). In nature this is not generally true because the temperature alone fluctuates in the course of the time so much that the rate of metabolism in poikilotherms changes considerably. This change may affect the rate of feeding as well as the excretion rate.

In the calculations of the feeding rate of bluegill, all possible factors affecting intake, body burden and excretion were taken into consideration. Three factors that are possible sources of errors in this study were (1) relationship between the biological half-life of cesium and the temperature; this was assumed to follow a Q_{10} relationship, (2) the limited number of ^{137}Cs analyses in different food items and stomach contents, and (3) the assimilation percentage of the minor food items of bluegill.

The radioisotopic method for studying the food consumption of a species that feeds on several food items is laborious because it requires several experiments to determine the assimilation percentages and many samples of food organisms to determine the possible seasonal cycling of the radioisotope concentration. Radiocesium, even though it has a high assimilation percentage and a long physical half-life, is not a convenient isotope for determining the food consumption rate in fish. The concentration of ^{137}Cs was not in equilibrium even in fish that were living in an environment such as White Oak Lake where the concentration of ^{137}Cs in water did not fluctuate considerably. If the concentration of the radioisotope fluctuates seasonally, the sampling has to be carried

on over one year to determine the amplitude of the seasonal cycling and the timing of the minimum and maximum concentration in fish.

The annual mean of daily feeding rates of bluegills determined by the balances of stable cesium and potassium was somewhat higher than the annual mean of daily feeding rates determined by ^{137}Cs balance. The concentrations of stable cesium and potassium were analyzed only in one composite stomach sample and the T_b of potassium was determined only in one group of fish. Therefore, the determination of the feeding rate based on the balance of ^{137}Cs was considered the most accurate. The use of stomach samples to determine the concentrations of ^{137}Cs, stable cesium and potassium in food may produce underestimates because the assimilation of these elements from food may be faster than the rate of digestion and passage of food through the stomach. This source of error was not included in the calculation of the feeding rate using ^{137}Cs balance because the concentration of ^{137}Cs in diet was calculated on the basis of its concentrations in different food items.

The most practical method would be to use a radioisotope whose concentration is in equilibrium in the fish or a stable element whose concentration is in equilibrium. Potassium would be one possible stable element that could be used for determining the feeding rates of fish in nature. As shown here, it is possible to use ^{42}K to determine the biological half-life of potassium in fish (see pp. 62-63). This method can be made even more accurate by separating ^{42}K chemically from water, which increases its counting efficiency in two ways, viz, by (1) counting ^{42}K as β-emitter (only 11% of the desintegrations of ^{42}K

produce γ-quanta compared to 100% β-emission) and (2) by decreasing the volume of sample when the geometry becomes better for counting radioactivity.

Feeding Rates of Bluegill in White Oak Lake

There was a positive relationship between temperature and feeding rate of bluegill in White Oak Lake (p. 69). The feeding rate was at a minimum in February and increased slowly during March and April. During May the feeding increased rapidly towards a maximum in June after which time there was a gradual decrease until February. The feeding rate of the White Oak Lake bluegill did not decrease in the fall of 1967 as rapidly as in the fall of 1968. This difference might have been caused by the differences in the temperature between these two summers. The summer of 1967 was cold while the summer of 1968 was hot (Fig. 10, p. 70). Seasonal patterns of bluegill feeding rates differed somewhat from the feeding rates of bluegill in two lakes in Minnesota where the volume of stomach contents decreased rapidly after the spring maximum (Seaborg and Moyle, 1964).

It has been found that the feeding rate of fish increases with temperature until a certain optimum temperature is reached, after which the feeding rate again decreases (Pentelow, 1939; Ricker, 1946; Baldwin, 1957; Kinne, 1960; Warren and Davis, 1967). Considering this fact and the seasonal adaptation processes, it is apparent that the feeding rate is only partly correlated with temperature on a seasonal basis. However, annual differences in water temperature caused by weather or thermal pollution may cause differences in the annual

feeding rates of fish. These variations in annual feeding rates may be completely correlated with the annual temperature variations if the extremes fall within the biokinetic range of the species.

The maximum daily meal of bluegill of age-group IV in White Oak Lake was 3.2% of their body weight. This was somewhat lower than the maximum daily meal (3.6%) of bluegills in laboratory conditions (Gerking, 1962). The daily meal of White Oak Lake bluegills in June was close to that in Lake Opinicon, Ontario, where it was 2.5% in early summer (Keast and Welsh, 1968). Seaborg and Moyle (1964) determined the daily meal of bluegill to be 1 - 2% for the whole summer in Minnesota.

The ecological growth efficiency (Gerking, 1962) in age-group IV bluegill in White Oak Lake was 4.2% for the period from April to October, while it was 9% for age-groups II - IV in Wyland Lake, Minnesota, for the period from May to October (Gerking, 1962). In Wyland Lake, growth efficiency of age-group IV bluegills was 6.5% for the summer months. Bluegill growth in White Oak Lake was faster in young fish than in Wyland Lake bluegill, but the gain in weight of age-group IV was greater in Wyland Lake bluegill (31.5 g) than in the White Oak Lake bluegill (18.9 g). The low growth efficiency of the age-group IV bluegills in White Oak Lake is probably caused by the declining growth rate at that age (see Table 3, p. 24).

CHAPTER VII

SUMMARY

The concentration of ^{137}Cs in bluegill increased linearly with the size of bluegill up to 70 g after which the concentration reached a steady state with a considerable seasonal cycling. The maximum concentration was in February and the minimum in August. The concentration of stable cesium, however, did not increase with the fish size, but followed a seasonal cycle similar to that of the ^{137}Cs concentration. Potassium concentrations were constant throughout the year and were considered to be in an equilibrium state. The body burdens of these elements in bluegill, including potassium, were not in equilibrium because fish grew every year, and consequently the body burden was increasing.

The biological half-life of ^{137}Cs in bluegill consisted of two components. The short component (T_{b_1}) was 7.6 d and the long component (T_{b_2}) 187 d at the mean annual temperature of White Oak Lake (15.8 C). The proportions of assimilated ^{137}Cs after a single feeding excreted by T_{b_1} and T_{b_2} were 37% and 63%, respectively. The biological half-life of ^{137}Cs in bluegill increased with the weight. The biological half-life of potassium consisted only of one component. It was 40 d at 15.8 C in bluegills of 70 g.

The assimilation of ^{137}Cs increased also with the fish size.. It was 7% in fish of 10 g and 16% in fish of 90 g which were fed Chironomus larvae that

were raised in White Oak Lake sediments. The assimilation of ^{137}Cs was about 70% from food not containing clay particles. Only 3% of ^{137}Cs in detritus was assimilated by large bluegill. The assimilation of ^{42}K was 98% from White Oak Lake Chironomus larvae and 72% from White Oak Lake detritus.

The minimum ^{137}Cs intake occurred in March and the maximum in September while stable cesium intake was greatest in January and least in August. The fluctuation of ^{137}Cs intake was caused by the fluctuation in concentration of this isotope in bluegill's food and the seasonal fluctuation of food consumption rates. Fluctuations in stable cesium intake were probably caused by similar factors. Potassium intake followed closely the fluctuation of the feeding rate, being maximum in July and minimum in February.

Food consumption rates were positively correlated with water temperature except at temperatures above 25 C and in autumn after a hot summer. The daily meal was 0.8% of body weight in February and 3.2% in June, being 1.75% for the whole year.

The concentration of ^{137}Cs in White Oak Lake fish was the highest in golden shiner, gizzard shad, and largemouth bass while redear sunfish had the lowest concentration. Concentrations of ^{137}Cs in all species followed a seasonal cycling similar to that in bluegill.

The concentration of stable cesium was not correlated with the concentration of ^{137}Cs in different species of fish but was highest in goldfish and lowest in bluegill. Potassium concentrations did not fluctuate seasonally and were on the same level in all species.

The specific activity of ^{137}Cs in bluegill fluctuated from 3800 pCi/μg stable cesium in January to 5200 pCi/μg stable cesium in July while the specific activity of water was 2050 pCi/μg stable cesium. The specific activity varied considerably among different species. It was highest in golden shiner (5200 pCi/μg Cs) and lowest in goldfish (2000 pCi/μg Cs). These differences mean that there were differences in the behavior of ^{137}Cs and stable cesium in White Oak Lake. Therefore, the specific activity of water was not valid for predicting the concentration of ^{137}Cs in different species of White Oak Lake fish.

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